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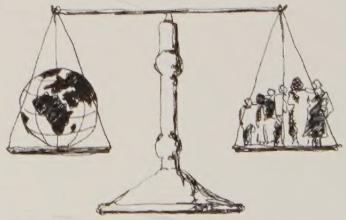

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Frequent cultivation prior to planting to prevent weed competition results in an opportunity for the use of arbuscular mycorrhizal fungus inoculum

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Abstract

Inoculation with arbuscular mycorrhizal (AM) fungi is a potentially useful tool in agricultural systems with limited options regarding use of synthetic chemicals for fertility and pest control. We tested the response of *Allium porrum* cv. Lancelot to inoculation with AM fungi in a field high in available P (169 $\mu g\, g^{-1}$ soil) that had been repeatedly cultivated to control weeds. Seedlings were inoculated during the greenhouse production period with a mixed species inoculum produced on-farm in a compost and vermiculite medium with *Paspalum notatum* Flugge as a nurse host. Inoculated and uninoculated seedlings were the same size at outplanting. Inoculated seedlings were over 2.5-fold greater in shoot weight and shoot P content than uninoculated seedlings at harvest. These results demonstrate the potential yield benefits from inoculation with AM fungi in situations where farm management practices may negatively impact on indigenous populations of AM fungi.

Key words: leek, stale seedbed

Introduction

Arbuscular mycorrhizal (AM) fungi are soil fungi that form a mutualistic symbiosis with the majority of crop plants[1]. Although the symbiosis has been shown to enhance water relations and disease resistance of the host plant, the most commonly measured benefit the plant receives from the symbiosis is enhanced mineral nutrient, notably P, uptake. The extra-radical phase of the fungus (ERM) acts in effect as an extension of the root system for the uptake of nutrients that are immobile in the soil solution. These nutrients are available to non-mycorrhizal roots only from the volume of soil explored by root hairs, approximately a 1–2 mm distance from the root. The ERM may extend up to 14 cm from the root, thereby exploring a greater soil volume for P[2]. This is reflected in the soil P levels required for optimal growth of non-mycorrhizal *Allium porrum* being much higher than that of mycorrhizal *A. porrum*, i.e., 80–100 versus 15–25 $\mu g\, P\, g^{-1}$ soil, respectively[3].

Utilization of the AM symbiosis is most attractive and perhaps essential in organic farming, where application of synthetic chemicals for fertilization and pest control are prohibited. Two options are available to farmers: (1) better utilize the AM fungi indigenous to their soils through management techniques that bolster the size and functioning of these populations; or (2) utilize inocula of AM fungi[4]. Management options that enhance mycorrhizal functioning include reduced tillage[5], overwintering cover crops[6,7] and crop rotation[8]. Two options for the use of inoculum are to use commercially available inocula or for the farmer to produce it on-farm[9–11].

One significant challenge that all farmers face is weed control. This is particularly challenging for organic farmers due to the constraints against the use of synthetic chemicals. An alternative to labor-intensive methods for such farmers is mechanical cultivation prior to planting to produce a 'stale seedbed'[12,13]. This is a cycle of cultivation,

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germination and early growth of weed seedlings, recultivation to uproot and desiccate those seedlings, and germination of yet more weed seeds. The cycle is repeated until the weed seed population in the upper layer of soil has been depleted.

Tillage operations by farmers are known to impact on the AM fungus community[14]. The ERM has dual functions as both the nutrient-absorbing organ of the symbiosis and as infective propagules. Soil disturbance has been shown to decrease the AM fungus colonization of seedlings germinating in the disturbed/tilled soil and decrease the uptake of P into the shoots of the seedlings[15,16]. Therefore, soils in which the stale seedbed method has been used for weed control may be strong candidates for utilization of AM fungus inoculum. We conducted an experiment to test the utilization of AM fungus inoculum produced on-farm in a mixture of compost and vermiculite to grow *A. porrum* L. seedlings in a field that had been repetitively cultivated prior to planting to control weeds.

Materials and Methods

Inoculum production

Inoculum of AM fungi was produced on-farm at the Rodale Institute, Kutztown, PA, USA during the 2008 growing season[17]. Briefly, a 1:4 [v/v] mixture of yard clippings compost and vermiculite, respectively, was prepared and placed into 7 gallon black plastic bags ('Grow Bags', Worm's Way, Bloomington, IN 47404, USA). *Paspalum notatum* Flugge seedlings, colonized by one of the following AM fungi: *Glomus intraradices* Schenck & Smith (DAOM 181602); *Glomus geosporum* (Nicolson & Gerdemann) Walker, *Glomus etunicatum* Becker & Gerdemann, *Glomus claroideum* Schenck & Smith and *Glomus mosseae* (Nicol. & Gerd.) Gerdemann & Trappe, originally isolated from the Rodale Institute Experimental Farm, Kutztown, PA, USA; and *Glomus* sp., isolated from the Stoneleigh Estate, Villanova, PA, USA were transplanted into the bags on June 18, 2008. Bags were weeded and watered as needed throughout the growing season. The *P. notatum* host plants were killed by cold winter temperatures, and the AM fungi overwintered outdoors in the growth medium.

Compost and vermiculite mixture from bags containing the above-listed AM fungi was collected on March 17, 2009. A most probable number bioassay conducted on the mixed inoculum, utilizing three replicate pots per dilution and tenfold dilutions ranging from 10^{-1} to 10^{-5}, indicated a propagule density of 120 cm^{-3}.[18]

Production of A. porrum *seedlings*

Seeds of *A. porrum* L. cv. Lancelot (Seedway LLC, 99 Industrial Pike, Elizabethtown, PA 17022, USA) were sown into 98-cell plastic flats (25 cm^3 $cell^{-1}$) on March 20, 2008. The flats contained either a 1:4 (v/v) mixture of the mixed species AM fungus inoculum and horticultural peat-based potting media (Berger BM6, Saint-Modeste, Quebec G0L 3W0), respectively, or straight potting media for the uninoculated treatment. Flats were placed in a growth chamber under controlled conditions (16 h/8 h, 25°C/18°C day/night) for 1 month, after which they were moved to a greenhouse. After seedling emergence, flats of mycorrhizal plants received a weekly watering with Hoagland's nutrient solution without P while the uninoculated flats received Hoagland's solution modified to contain 0.01 mM KH_2PO_4[19]. These fertilization regimes were designed to promote AM fungus colonization of inoculated plants and provide sufficient P to the uninoculated plants.

Site preparation and outplanting

Field aspects of the experiment were conducted at a farm in Lititz, PA, USA. The soil was a Berks silt loam (loamy-skeletal, mixed, active, mesic Typic Dystrudepts) with pH = 7.0 and available P = 169 $\mu g\, g^{-1}$ soil (Mehlich 3). The field plot had been used to grow cantaloupes (*Cucumis melo*) in 2008, and there was no overwintering cover crop utilized prior to the 2009 growing season. Grass clippings and leaf compost was added at the rate of 22.4×10^3 kg ha^{-1}. The site was roto-tilled three times to control weeds prior to outplanting the seedlings on July 17, 2009. Soil collected to a depth of 20 cm from five locations within the plot on the day of planting was pooled (approximately 1.5 liter total) and utilized for a most probable number bioassay to measure the status of the indigenous population of AM fungi. The assay indicated that the soil contained 2.2 propagules of AM fungi cm^{-3}.

The seedlings were outplanted in two rows 90 cm apart with 23 cm between plants within the row. The two treatments were represented in alternating runs of 15 plants; i.e., 15 inoculated plants, followed by 15 uninoculated plants, etc., with ten such 15 plant units per inoculation treatment.

Data collection and analysis

Six plants of each treatment were withheld from outplanting for characterization of plant growth and mycorrhizal status at time zero. Shoot length and dry weight were recorded. Shoot P concentration was determined colorimetrically after H_2SO_4 plus H_2O_2 digestion[20]. Percentage root length colonized by AM fungi was determined at 20 × magnification under a dissecting microscope via the gridline intersect method after clearing and staining the roots[21,22].

Ten plants per treatment were harvested from the field on September 24, 2009. In addition to the data collected at outplanting, the diameters of the freshly harvested stems were measured at their widest point.

Data were analyzed using analysis of variance. Measurements for which significant treatment effects were found were characterized further using Tukey's method of multiple comparisons ($\alpha = 0.05$).

Table 1. Status of *A. porrum* cv. Lancelot at time of outplanting[1].

Treatment	Height (cm)	Shoot dry wt (g)	Shoot P (% dry wt)	Shoot P Total (mg)	Colonization (% root length)
Non-myc	21.1 ± 1.4	0.047 ± 0.009	0.140 ± 0.016	0.059 ± 0.009	0.0
Mycorrhizal	22.4 ± 2.0	0.043 ± 0.008	0.171 ± 0.018	0.068 ± 0.010	43.0 ± 11.9
Pr>F	0.5936	0.7553	0.2378	0.5047	0.0046
MSD[2]	5.5	0.0276	0.0455	0.030	26.4

[1] Means of six observations ± SEM. Non-myc, non-mycorrhizal.
[2] Minimum significant difference from Tukey's test ($\alpha = 0.05$).

Table 2. Status of *A. porrum* cv. Lancelot at harvest[1].

Treatment	Height (cm)	Shoot P (% dry wt)	Shoot P Total (mg)	Colonization (% root length)
Non-myc	71.8 ± 1.8	0.389 ± 0.013	29.9 ± 3.8	42.5 ± 5.2
Mycorrhizal	83.2 ± 1.8	0.415 ± 0.011	83.7 ± 6.8	53.1 ± 6.6
Pr>F	0.0002	0.1449	< 0.001	0.2222
MSD[2]	5.2	0.036	16.4	17.6

[1] Means of ten observations ± SEM. Non-myc, non-mycorrhizal.
[2] Minimum significant difference from Tukey's test ($\alpha = 0.05$).

Results

An important consideration in work with AM fungi is that inoculated and uninoculated plants be of the same size and P status at the time of outplanting or imposition of experimental treatments. There were no significant differences between treatments at the time of outplanting (Table 1). Inoculated seedlings were well colonized by AM fungi. No colonization was found in the uninoculated plants.

Aboveground growth differences were striking at harvest. Shoot biomass of inoculated plants was 266% and stem diameter 159% that of uninoculated plants (Fig. 1). Uninoculated plants had become colonized by indigenous AM fungi to the point that percentage root length colonized was not significantly different from that of the inoculated plants (Pr>F = 0.2222) after 10 weeks in the field (Table 2). Phosphorus content of shoots of inoculated plants was 280% of that of uninoculated plants.

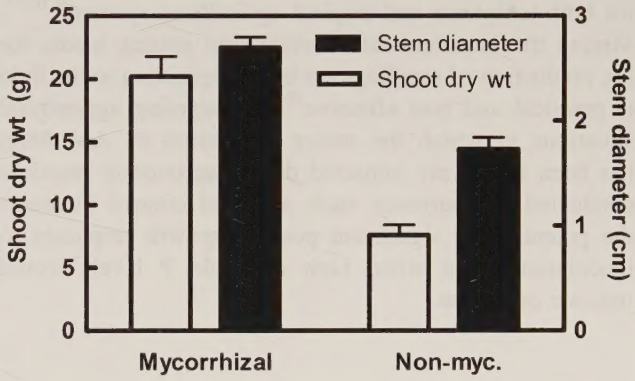

Figure 1. Stem diameter and shoot dry weight of *A. porrum* cv. Lancelot at harvest. Means of ten observations ± SEM. Both measurements were significantly different between treatments (Pr>F ≤ 0.0001). Non-myc, non-mycorrhizal.

Discussion

Inoculation of *A. porrum* seedlings with AM fungi during the greenhouse growth phase significantly increased their growth in the field despite an available P level of 169 µg P g^{-1} soil.

Two soil characteristics are primary determinants of the potential for a response to inoculation with AM fungi. The first is the level of available P in the soil. High levels of available P increase root P concentrations, which result in lower colonization of roots by AM fungi[23]. The level of available P above which there may not be a response to mycorrhization varies with the soil type and host plant, and has been shown to range from 50 to 140 µg P g^{-1} soil[24,25]. No growth response at final harvest was seen by Sorensen et al.[7] for *A. porrum* in soil with moderate available P levels, ranging from 26 to 50 µg P g^{-1} soil in the presence of the healthy AM fungus community in the soils of their experiment.

The second soil factor exerting control over the potential for a response to inoculation with AM fungi is biological: the abundance of the indigenous population with which the inoculated AM fungi must compete[9,26]. No data are available to form a soil inoculum potential versus growth response curve specifically for *A. porrum*. Sieverding[9] found that *Manihot esculenta* was unlikely to respond to inoculation when the indigenous AM fungus population

exceeded 12 propagules g^{-1} soil. Here, we had a lower population of AM fungi (2 propagules cm^{-3}) than that, and saw a significant growth response. Even at low levels, the indigenous AM fungus population produced colonization in the uninoculated plants that was statistically equivalent to that of the inoculated seedlings by the end of the experiment (Table 2). The growth benefit from inoculation came from the advantage to seedling growth conferred by outplanting with an already-established symbiosis.

In addition to repeated tillage/cultivation, other agricultural practices negatively impact on the indigenous AM fungus community and therefore create opportunities for the use of AM fungus inoculum. Bare fallows deprive AM fungi of host roots to colonize, resulting in a decline in vigor and infectivity[27]. Non-host overwintering cover crops such as *Brassica* sp. not only likewise deprive AM fungi of fixed carbon but also produce compounds inhibitory to the growth of AM fungus hyphae[28]. Continuous monocultures of commercially attractive crops select for those members of the AM fungus community best at reproducing with those host plants, not necessarily the most effective symbionts[8]. Lastly, fertilization may select for ineffective AM fungi[29]. These situations produce opportunities for inoculation with effective isolates of AM fungi[4].

On-farm production of AM fungi is a viable technology for both temperate and tropical agricultural systems[9,10,17]. Mixing the inoculum into horticultural potting media for the production of seedlings for later outplanting to the field is practical and cost effective[30–32]. Targeting agricultural situations in which the native population of AM fungi has been negatively impacted due to agronomic practices conducted for purposes such as weed control increases the potential for significant positive growth responses to inoculation, even when high available P levels would indicate otherwise.

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Water infiltration and surface-soil structural properties as influenced by animal traffic in the Southern Piedmont USA

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Abstract

Surface-soil structural condition in perennial pastures is expected to be modified by how forage is (a) harvested through haying or grazing and (b) stimulated through source of nutrients applied, as well as by compactive forces, e.g., grazing cattle or hay harvest machinery. Changes in surface-soil condition can affect hydrologic processes that have important implications for plant growth, greenhouse gas emissions and off-site water quality. We determined the effects of harvest management and nutrient source on the rate of ponded water infiltration and penetration resistance in a bermudagrass [*Cynodon dactylon* (L.) Pers.]/tall fescue (*Lolium arundinaceum* Schreb. S.J. Darbyshire) pasture on a Typic Kanhapludult in Georgia. During a period when soil was wet (61% water-filled pore space), the rate of water infiltration was 2.8 ± 1.5 times greater when forage was left unharvested as when hayed or grazed (mean ± standard deviation among nine nutrient source × harvest management comparisons). During a subsequent period, when soil was dry (28% water-filled pore space), the rate of water infiltration followed the same treatment pattern, but was not statistically different among harvest-management practices (1.5 ± 0.4 times greater between unharvested and other systems). Penetration resistance of the surface at 10 cm depth followed the order: unharvested (62 J) < hayed (100 J) < low grazing pressure (119 J) < high grazing pressure (137 J). Water infiltration during the wet period was negatively related ($P \leqslant 0.01$) to soil-water content ($r = -0.57$), penetration resistance at 0–10 cm depth ($r = -0.50$) and bulk density at 3–6 cm depth ($r = -0.53$), but was positively related to surface residue C ($r = 0.47$) and soil organic C concentration at 12–20 cm depth ($r = 0.42$). These results suggest that complex soil physical (i.e., aggregation, penetration resistance and infiltration) and biological (i.e., plant growth, surface residues and soil organic matter) interactions occur in pastures. We conclude that well-managed grazing systems with excellent ground cover should have adequate hydrologic condition to promote pasture productivity and limit environmental contamination from runoff. Further work is needed to understand the linkages between field- and watershed-scale hydrology in perennial pastures and their implications on water quality.

Key words: bulk density, compaction, pasture management, penetration resistance, soil organic carbon, surface residue

Introduction

Environmental quality and sustainability of agricultural systems are highly dependent on adequate soil functions. Some critical soil functions are to receive and recycle water, to support vigorous plant communities and to cycle nutrients without loss to the environment. Surface-soil characteristics are of particular importance, not only in affecting the environmental quality of a location but also for determining the environmental quality of neighboring ecosystems if runoff and gaseous emissions are high. Surface cover is an important determinant of how rainfall is partitioned into infiltration and runoff[1]. Humid-zone pastures generally have excellent surface cover, but still may have significant water runoff due to high soil water content or surface compaction with traffic from grazing animals (present whether soil is dry or wet). Since pastures are seldom tilled, nutrients and organic matter can accumulate at the soil surface, and therefore, runoff from pastures can be a threat to water quality[2]. However, high water infiltration allows nutrients accumulating at the surface of undisturbed soil to percolate into the rooting zone of pasture plants for efficient utilization and recycling. Surprisingly, relatively few data are available to assess the impact of how forage management might alter water infiltration in soils of the warm, humid region of the southeastern USA.

When forage is reduced in vigor due to high grazing pressure, the protective plant cover thins and various sized

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patches of bare ground are exposed, leading to low residue cover, compaction, and surface sealing during drying and wetting events, thereby reducing water infiltration and increasing runoff. In Texas, on a silty clay soil, the amount of surface ground cover (both living and dead organic material) was an excellent predictor of water infiltration rate[3]. In Pennsylvania, on clay loam and sandy loam soils, water infiltration was also positively related to surface ground cover and negatively related to cattle grazing intensity[4]. In a review of how animal grazing affects water infiltration, Greenwood and McKenzie[1] cited several studies documenting how moderate to heavy grazing pressure reduced water infiltration compared to ungrazed or lightly grazed treatments. Trimble and Mendel[5] stated: 'on uplands, heavy grazing compacts the soil, reduces infiltration, increases runoff, and increases erosion and sediment yield'. Clearly there is a need to quantify pasture-surface condition, whether forage is grazed or not, when assessing water infiltration.

The impact of grazing on surface-soil structural condition has not been evaluated in great detail in the southeastern USA. The results of the few studies are reviewed here. In a bermudagrass/tall fescue hayfield in the Piedmont of Georgia, water runoff volume, time to runoff and nutrient loss were not statistically affected whether surface soil was aerated or not[6]. In a grazed bermudagrass/tall fescue pasture in the Piedmont of Georgia, water runoff volume (13% of rainfall) was not affected whether pastures were grazed continuously or rotationally (3 days grazed and 21 days rested)[7]. During a total of 4 years of this same study, water runoff volume was $15 \pm 4\%$ of rainfall that occurred in 23 ± 5 events per year[8].

We have been investigating the effects of harvest and nutrient-source treatments on various soil, plant and animal responses in the Piedmont of Georgia to understand the relationships between productivity and conservation. During the first 4 years, we found that soil bulk density was lower in the 0–2 cm depth when grazed than when ungrazed, but was greater in the 2–4 cm depth when grazed than when ungrazed, resulting in no difference when summed for the 0–6 cm depth[9]. We continued the evaluation of this experiment for 12 years and hypothesized that surface-soil structural conditions could have been altered by the presence or absence of grazing cattle. Our objective was to evaluate water infiltration and penetration resistance as indicators of animal impact on surface-soil properties and compare these indicators with associated changes in bulk density, soil organic C, surface residue C and N, and ground cover.

Materials and Methods

Site characteristics

A 15-ha upland field (33°22′N, 83°24′W) in the Greenbrier Creek subwatershed of the Oconee River watershed near Farmington, GA had previously been conventionally cultivated with various row crops for several decades prior to grassland establishment by sprigging of 'Coastal' bermudagrass in 1991. From 1994 to the end of summer in 1998, bermudagrass was the dominant forage[10]. 'Georgia 5' tall fescue was drilled (approx. 25 kg pure live seed ha^{-1}) directly into existing bermudagrass sod during November 1998, 1999 and 2000. Abnormally dry winter conditions prevented adequate establishment in 1998 and 1999, resulting in the need for repeated sowing. By 2005, tall fescue was $\geqslant 50\%$ of the botanical composition in all treatments. Long-term mean annual temperature in the area is 16.5°C, rainfall was 1263 mm and potential evapotranspiration was 1029 mm, based on the Thornthwaite equation. Dominant soils at the site are Madison, Cecil and Pacolet sandy loam (fine, kaolinitic and thermic Typic Kanhapludults).

Experimental design

The experimental design was a randomized, complete block with treatments in a split-plot arrangement in each of three blocks, which were delineated by landscape features (i.e., slight, moderate and severe erosion classes). Main plots were nutrient source ($n = 3$) and split-plots were harvest management ($n = 4$) for a total of 36 experimental units. Grazed paddocks were 0.69 ± 0.03 ha. Spatial design of paddocks minimized runoff contamination and facilitated handling of cattle (*Bos taurus*) through a central roadway. Each paddock contained a 3×4 m shade, mineral feeder and water trough placed in a line 15-m long at the highest elevation. Unharvested and hayed exclosures ($100\ m^2$) were randomly placed side-by-side in paired low- and high-grazing pressure paddocks of each nutrient source at the initiation of the study in 1994.

Nutrient-source treatments were: (1) inorganic only; (2) organic + inorganic mixture; and (3) organic only. From 1994 to 1998, nutrient-source treatments were: (1) inorganic fertilizer as NH_4NO_3 broadcast in May and July; (2) crimson clover (*Trifolium incarnatum* L.) cover crop + inorganic fertilizer (half of N assumed fixed and released by clover cover crop during spring and the other half as NH_4NO_3 broadcast in July); and (3) chicken (*Gallus gallus*) broiler litter ($5.4\ Mg\ ha^{-1}\ yr^{-1}$) broadcast in May and July. Nutrient-source treatments were modified after the first 5 years of management. Fertilizer application was targeted to supply $200\ kg\ N\ ha^{-1}\ yr^{-1}$ during the first 5 years and targeted to supply $270\ kg\ N\ ha^{-1}\ yr^{-1}$ during the next 7 years (see Franzluebbers and Stuedemann[11,12] for application details). From 1999 to the end of summer 2005, the three nutrient-source treatments were: (1) inorganic fertilizer as NH_4NO_3 broadcast in three applications in February–April, May–July and September–November; (2) single application of broiler litter ($\sim 2.7\ Mg\ ha^{-1}\ yr^{-1}$) broadcast in February–April and supplemented with inorganic fertilizer as NH_4NO_3 broadcast in May–July and September–November; and (3) multiple applications of

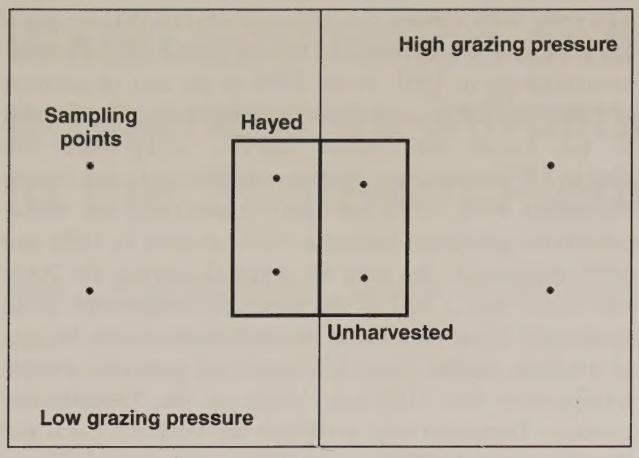

Figure 1. Schematic diagram showing relative location of infiltration rings and penetration resistance sampling points in one nutrient-source treatment that contained hayed and unharvested exclosures and adjacent low and high grazing pressure paddocks.

broiler litter broadcast three times in February–April, May–July and September–November (~8.1 Mg ha^{-1} yr^{-1}).

Harvest management regime consisted of: (1) unharvested biomass (cut and left in place at the end of growing season during years 1–5 and left unmanaged during years 6–12, except for an occasional woody plant removal); (2) low grazing pressure targeted to maintain 3.0 Mg ha^{-1} of standing forage after grazing; (3) high grazing pressure targeted to maintain 1.5 Mg ha^{-1} of standing forage after grazing; and (4) hayed monthly to remove above-ground biomass at 5-cm height. Yearling Angus steers grazed paddocks during a 140-day period from mid-May until early October during years 1–5 (mean body weight of 212 kg and mean stocking density of 5.8 and 8.7 steers ha^{-1} in low and high grazing pressure treatments, respectively). Grazing was extended into spring (March to May) and autumn (mid-October to early January) during years 6–12 with the presence of tall fescue. Grazing did not typically occur from mid-January to mid-March.

Sampling and analyses

Single-ring water infiltration[13] was determined during two separate sampling events in March and April 2006 from duplicate 30-cm-diameter steel rings placed approximately 7 m apart within an experimental unit (Fig. 1). Since both hayed and unharvested exclosures were adjacent to each other, distance between rings was also about 7 m apart between these treatments. Rings in adjacent grazed paddocks were approximately 15 m from rings in hayed and unharvested exclosures. This arrangement minimized some soil variations that were expected within large grazed paddocks (e.g., soil texture, cattle paths, etc.). Water (with blue dye added for better visualization) was delivered to rings via a Mariotti siphon system using a 20-liter graduated vessel to monitor quantity[13]. Volume readings were recorded at 10-min intervals for 1 h. A water head of approximately 5 cm was maintained inside each ring. Recharge of vessel with additional water was needed periodically. From cumulative water infiltrated from 10 to 60 min, we found that a linear regression provided a best fit in most cases ($r^2 = 0.97 \pm 0.01$ among estimates in March; $r^2 = 0.99 \pm 0.01$ among estimates in April), and therefore, we determined the intercept [designated as macropore filling (mm); functionally derived as the rapid entry of water filling open pores in surface soil, as well as saturation of large matric potential gradient during the first 10 min; analogous to the expression of sorptivity (mm min$^{-\frac{1}{2}}$)[13]] and slope [designated as infiltration rate (mm min^{-1})] for each measurement event [$n = 144$ (3 nutrient sources × 4 harvest management regimes × 3 replications × 2 subsamples × 2 periods)]. No prewetting of soil occurred, and therefore, soil with low antecedent moisture would be expected to have greater initial rate of infiltration due to macropore filling, but similar steady-state rate of infiltration as wet soil. Again, 'macropore filling' encompasses a combination of filling large voids in the soil matrix and saturating capillarity towards drier soil in the near surface.

Small, in-field runoff collectors (0.3 × 0.76 m)[14] were installed only in low grazing pressure paddocks with inorganic only and organic only nutrient sources for analysis of runoff occurrence from December 1999 to December 2004. A total of 20 runoff collectors in six paddocks were used to determine occurrence of runoff (0 = no runoff; 1 = runoff). Rainfall was recorded on-site with a data logger. A rainfall event was defined as daily rainfall >1 mm and ending whenever rainfall did not occur after 08 : 00 h the following morning. Rainfall events lasted 1–3 days. Frequency distribution of rainfall events was determined from categories of 1–10, 10–20, 20–30, 30–40, 40–50 and >50 mm. Occurrence of runoff was tested for significance among rainfall event classes and between nutrient-source treatments from the likelihood-ratio statistic using Type 3 source of error.

Penetration resistance was determined in March 2006 at locations ~1 m from infiltration rings. An impact penetrometer with a 2-kg hammer was dropped 0.74-m distance repeatedly onto a 2.03-cm-diameter cone with a 30° tip[15]. The number of strikes required to reach a depth of 10, 20 and 30 cm was recorded. Each strike contained the equivalent kinetic energy of 14.5 J. Soil water content was determined at the same time at 0–20-cm depth with time-domain reflectrometry (Field Scout TDR-300, Spectrum Technologies Inc., Plainfield, IL).

Surface residue and soils were sampled in January/February 2006 and results were presented in detail elsewhere[16]. Surface residue was a composite of eight 0.04-m^2 areas randomly selected within each of three zones within grazed paddocks (i.e., 0–30, 30–70 and 70–100 m distances from livestock shades) and from one zone in each exclosure. Following the removal of vegetation above a height of ~4 cm, surface residue, including plant stubble, was cut to the mineral surface with battery-powered hand

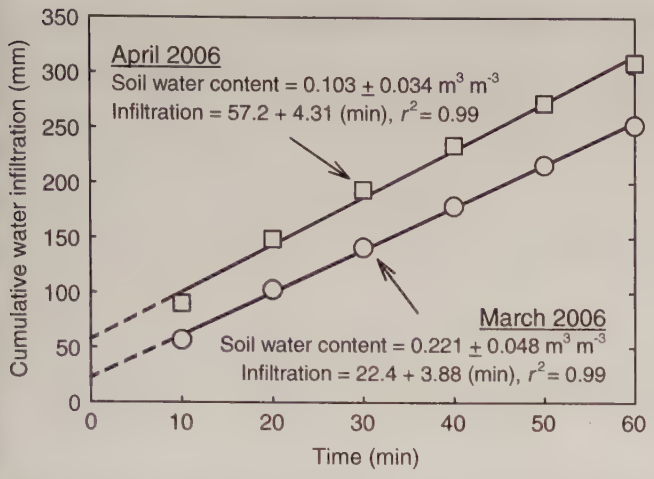

Figure 2. Mean water infiltration across treatments and replicates during the March and April sampling periods at the grassland site near Farmington, GA.

shears, bagged, dried at 55°C for several days and ground to < 1 mm. A single 4-cm-diameter soil core was collected from under each of the eight residue sampling sites and composited. Soil bulk density was calculated from the oven-dried soil weight (55°C, 72 h) and pooled-core volume (302, 302, 603 and 804 cm^3, respectively) from depths of 0–3, 3–6, 6–12, and 12–20 cm. Surface residue and soil were analyzed for total C and N with dry combustion.

Ground cover was determined in June 2005 from visual inspection of 0.25 m^2 areas by an experienced technician of the percent basal area (minimum of 5% units) of the categories, Coastal bermudagrass, common bermudagrass, tall fescue, weeds and bare ground. Estimates were made from 30 locations per experimental unit in grazed paddocks and 10 locations per experimental unit in exclosures.

Multiple responses within an experimental unit (e.g., two infiltration and penetration resistance estimates in each plot, three residue and bulk density estimates in grazed paddocks) were averaged prior to statistical analysis of the experiment as a split-block with four replications. Transformations were necessary to achieve normality in data distributions, e.g., macropore filling (mm) and penetration resistance (J) were log-transformed and infiltration rate (mm min^{-1}) and soil water content (m^3 m^{-3}) were square-root transformed. Analysis of variance was conducted with nutrient source as main plot ($n = 3$) and harvest management as split-plot ($n = 4$). *A priori* orthogonal contrasts were constructed to separate treatment effects. Significant differences among treatment means were declared at $P \leqslant 0.05$. Significant correlations among soil and water variables were declared at $P \leqslant 0.01$.

Results and Discussion

Cumulative water infiltration was nearly linear with time after the first 10 min (Fig. 2), supporting the simple approach to split the infiltration response into a macropore filling portion during the first 10 min and a steady-state linear rate of infiltration during the remainder of the hour of evaluation. Wetter antecedent soil condition in March resulted in somewhat lower infiltration than a month later, during which time tall fescue flourished with the springtime flush of growth, thereby effectively removing surface-soil moisture. Rate of water infiltration from 10 to 60 min was on average 0.4 mm min^{-1} greater (11%) during the drier period in April, but this was not considered significant based on least-square difference values reported in Table 1. When the rate of water infiltration was calculated using the 20–60 min period only, the April value was only 7% greater than the value in March. Macropore filling (intercept in Fig. 2, which accounted for the non-linear infiltration during the first 10 min) in the drier period in April was more than double that in March. The difference in macropore filling between March and April of 35 mm corresponded very well to the difference in soil-water content. With a mean bulk density of 1.35 Mg m^{-3} at a depth of 12 cm in 2006 (Table 2), water-filled pore space was 0.61 and 0.28 m^3 m^{-3} in March and April, respectively, suggesting that the additional 35 mm of macropore filling in April would have filled the surface 11 cm of soil.

Wet period in March 2006

Soil water content was unaffected by nutrient source and harvest management, although some trends emerged for lower water content with organic+inorganic nutrient source than with other nutrient sources ($P = 0.09$) and for greater water content in grazed than ungrazed pastures ($P = 0.12$) (Table 1). The trend for greater water content when pastures were grazed than ungrazed could have been due to greater ground cover (greater basal area that would limit evaporation) and soil organic C content (increasing water retention) (Table 2) in grazed treatments. Herbage covering the ground could have increased transpiration loss of water, but reduced evaporation. However, transpiration loss in this spring evaluation would have been low due to limited growth at the time of measurement. Macropore filling was also not affected by nutrient source or harvest management regime, suggesting that perennial pasture growth itself may have been more dominating than long-term application of manure on soil macropore development.

The rate of water infiltration was unaffected by nutrient source, but was significantly ($P < 0.001$) affected by harvest anagement regime (Table 1). The infiltration rate was greater when forage was unharvested (6.7 mm min^{-1}) than all other harvest management strategies (2.6–3.2 mm min^{-1}). An interactive trend ($P = 0.07$) also occurred, in which there was a larger difference in the infiltration rate between unharvested (9.3 mm min^{-1}) and other harvest management regimes (4.0 mm min^{-1}) using organic+inorganic nutrient source than between unharvested (5.4 mm min^{-1}) and other harvest management regimes (3.1 mm min^{-1}) using organic only nutrient source. Reduced

Table 1. Soil water content, macropore filling and linear water infiltration rate as affected by nutrient source and harvest management during a wet sampling period in March 2006 and a dry sampling period in April 2006.

		March 2006						April 2006		
Nutrient source (NS)	Harvest management (HM)	Soil water content ($m^3\ m^{-3}$)	Macropore filling (mm)	Infiltration rate ($mm\ min^{-1}$)	Penetration resistance (J)			Soil water content ($m^3\ m^{-3}$)	Macropore filling (mm)	Infiltration rate ($mm\ min^{-1}$)
					0–10 cm	10–20 cm	20–30 cm			
Inorganic only	Mean	0.235	22	2.1	107	139	147	0.104	51	3.9
Organic + inorganic	Mean	0.187	23	4.7	98	141	126	0.093	41	4.5
Organic only	Mean	0.235	16	3.3	96	147	147	0.105	52	3.6
Mean	Unharvested	0.201	21	6.1	62	127	135	0.088	59	5.0
Mean	Low grazing pressure	0.225	20	3.0	119	155	154	0.103	44	4.1
Mean	High grazing pressure	0.231	20	2.2	137	154	146	0.111	44	3.4
Mean	Hayed	0.217	19	2.4	100	135	125	0.103	45	3.4
Inorganic only	Unharvested	0.220	23	5.1	70	133	149	0.084	85	4.7
Inorganic only	Low grazing pressure	0.227	22	1.9	118	136	151	0.095	38	3.2
Inorganic only	High grazing pressure	0.264	22	0.8	155	164	163	0.129	42	4.5
Inorganic only	Hayed	0.231	19	1.5	101	126	125	0.113	52	3.2
Organic + inorganic	Unharvested	0.192	26	8.9	56	122	132	0.093	38	5.5
Organic + inorganic	Low grazing pressure	0.191	30	3.9	117	150	142	0.098	43	4.2
Organic + inorganic	High grazing pressure	0.192	14	4.6	145	166	141	0.087	42	4.0
Organic + inorganic	Hayed	0.195	25	2.6	99	130	97	0.092	40	4.5
Organic only	Unharvested	0.217	15	4.8	60	126	126	0.086	65	5.0
Organic only	Low grazing pressure	0.260	11	3.4	122	180	171	0.115	52	5.0
Organic only	High grazing pressure	0.238	28	2.1	114	136	137	0.120	48	2.1
Organic only	Hayed	0.227	14	3.4	101	151	162	0.103	44	2.7
Source of variation	df	----------	----------	----------	----------	Pr>F	----------	----------	----------	----------
NS1: inorganic versus mixed and organic only	1	0.26	0.72	0.14	0.16	0.68	0.26	0.74	0.29	0.86
NS2: organic only versus inorganic + organic	1	0.08	0.38	0.39	0.71	0.64	0.07	0.50	0.09	0.40
HM1: grazed versus ungrazed	1	0.12	0.92	0.02	< 0.001	0.004	0.05	0.18	0.37	0.54
HM2: unharvested versus hayed	1	0.31	0.79	< 0.001	< 0.001	0.38	0.44	0.22	0.27	0.14
HM3: low versus high grazing pressure	1	0.74	0.91	0.30	0.08	0.98	0.60	0.50	0.99	0.51
NS1 × HM1	1	0.97	0.88	0.38	0.89	0.77	0.97	0.92	0.17	0.71
NS1 × HM2	1	0.80	0.86	0.71	0.25	0.25	0.48	0.43	0.56	0.94
NS1 × HM3	1	0.20	0.94	0.46	0.25	0.09	0.37	0.18	0.78	0.19
NS2 × HM1	1	0.60	0.40	0.72	0.29	0.41	0.38	0.28	0.71	0.80
NS2 × HM2	1	0.65	0.94	0.07	0.81	0.50	0.03	0.54	0.45	0.55
NS2 × HM3	1	0.60	0.03	0.29	0.16	0.04	0.37	0.61	0.94	0.25

Table 2. Characteristics of surface soil as affected by harvest management (mean ± standard deviation among 3 replication × 3 nutrient-source treatments).

Harvest management	Ground cover (%)[1]	Surface residue C ($Mg\,ha^{-1}$)[2]	Soil organic C ($Mg\,ha^{-1}$)[3]	Bulk density ($Mg\,m^{-3}$)[3]
Unharvested	79 ± 11	4.0 ± 1.4	17.8 ± 2.3	1.36 ± 0.07
Low grazing pressure	99 ± 1	2.5 ± 0.9	21.4 ± 0.7	1.34 ± 0.03
High grazing pressure	98 ± 1	1.7 ± 0.3	21.4 ± 1.6	1.36 ± 0.05
Hayed	81 ± 6	1.9 ± 0.6	15.1 ± 1.0	1.36 ± 0.07

[1] Ground cover was estimated from basal area within 0.25 m^2 (30 subsamples per plot when grazed and 10 subsamples per plot when ungrazed) in June 2005.
[2] Sampled in January/February 2006 from 0.04 m^2 areas (eight subsamples per plot).
[3] Sampled in January/February 2006 from a 4-cm-diameter core at a depth of 0–12 cm (eight subsamples per plot).

infiltration in grazed compared with unharvested management was likely due to compression of soil at 3–6 cm depth (bulk density was 1.46 $Mg\,m^{-3}$ when grazed and 1.39 $Mg\,m^{-3}$ when unharvested)[16]. This result was consistent with the majority of literature that shows greater bulk density with increasing animal traffic[1]. The interactive trend between harvest and nutrient source may have been a result of the difference in soil water content between nutrient sources, in which soil water content was lowest when unharvested with organic+inorganic nutrient source (Table 1). As noted from the difference in mean water infiltration between the dry and wet periods, lower soil water content would lead to a greater rate of water infiltration.

Penetration resistance was affected mostly by harvest management and mostly in the surface soil at a depth of 0–10 cm, but some effects also occurred at lower depths (Table 1). At a depth of 0–10 cm, penetration resistance was not affected by nutrient source, but was greater ($P < 0.001$) when grazed (128 J) than when ungrazed (81 J), was greater ($P < 0.001$) when hayed (100 J) than when unharvested (62 J) and tended to be greater ($P = 0.08$) under high grazing pressure (137 J) than under low grazing pressure (119 J). Compared with no animal or tractor traffic when unharvested, mechanical removal of forage as hay reduced surface residue C (Table 2) and applied surface pressure with tractor wheel traffic [although harvest was with a small tractor of 800 kg with 18- (front) and 25-(back)cm-wide wheels]. These effects likely caused the difference in penetration resistance between hayed and unharvested treatments. Greater penetration resistance with grazing (Table 1), despite high surface residue C compared with haying (Table 2), suggests that animal traffic was a significant force that compressed surface soil. The trend for greater penetration resistance with high grazing pressure than with low grazing pressure was further evidence that animal traffic influenced soil firmness.

At a depth of 10–20 cm, penetration resistance was not affected by nutrient source, but was greater ($P = 0.004$) when grazed (154 J) than when ungrazed (131 J). Grazing pressure (i.e., low versus high grazing pressure) had an interactive effect with nutrient source ($P = 0.04$) at 10–20 cm; penetration resistance tended to be higher with high grazing pressure than with low grazing pressure using inorganic fertilizer, was similar between grazing pressures using organic+inorganic nutrient source, and was lower with high grazing pressure than with low grazing pressure using organic only nutrient source. This interaction of harvest management with nutrient source is curious and suggests that nutrient availability from organic fertilizer may have positively affected root growth and its influence on stabilizing soil structure. Further research is warranted to better understand this interaction.

At a depth of 20–30 cm, penetration resistance was not affected by nutrient source, but continued to be greater with grazed than with ungrazed systems (Table 1). A significant interaction occurred between nutrient source and harvest management, wherein penetration resistance was lower with haying than unharvested management using organic+inorganic nutrient source, but was greater with haying than unharvested management using organic only as nutrient source ($P = 0.03$).

Resistance of soil to root exploration can be a limitation to plant productivity. Point measurements of penetration resistance should be considered an index of soil resistance to root penetration only, since roots explore soil intra-aggregate pores of least resistance, while mechanical devices must penetrate whatever solids and voids are encountered in the insertion path. Soil penetration resistance is a function of bulk density and soil water content, i.e., as bulk density increases and soil water content decreases, penetration resistance increases[17]. Soil penetration resistance on silt loam and silty clay loam soils at a depth of 0–10 cm was occasionally greater with winter grazing of corn stalks in Iowa[18], as follows. Soil penetration resistance with grazing was 31 ± 9% ($n = 6$) greater than without grazing when soil was frozen for only 22 ± 33% of the time, but only 13 ± 6% greater (not significant) ($n = 9$) when soil was frozen for 72 ± 41% of the time[18].

Since soil water content is known to affect penetration resistance[19], we tested the relationship between soil water content and penetration resistance within each depth increment. Soil water content significantly influenced penetration resistance only at a depth of 0–10 cm (penetration resistance = 52 ± 249*soil water content, $r^2 = 0.11$,

$P = 0.004$), although there was a positive tendency for soil-water content to influence penetration resistance at lower depths also. Adjusting penetration resistance to the mean soil water content of $0.218\ m^3\ m^{-3}$ would have increased the unharvested mean by 4 J and the hayed mean by < 1 J and would have decreased the low grazing pressure mean by 2 J and the high grazing pressure mean by 3 J. None of these changes would have altered the significance or interpretation from the unadjusted means.

Dry period in April 2006

Soil water content was not affected by nutrient source or harvest management regime during the dry period of evaluation (Table 1). Macropore filling was only different ($P = 0.04$) between unharvested (73 mm) and hayed (45 mm) management and there were no significant interactions. This result indicates that macropores were more abundant when grass was completely undisturbed, but this result was inconsistent with the evaluation under wet conditions. The rate of water infiltration was also not affected by nutrient source or harvest management, although the strongest trend ($P = 0.11$) was between unharvested ($5.5\ mm\ min^{-1}$) and hayed management ($3.7\ mm\ min^{-1}$), which was consistent with the effect that occurred during the wetter period in March. The lack of treatment differences during the dry period suggest that soil structural condition may not have been limiting water infiltration at a stage when soil would be most vulnerable to the process of degradation, i.e., if greater runoff were to occur on soil with dry antecedent moisture, then the subsequent process of pasture degradation would certainly follow with lower productivity and loss of nutrients via runoff. However, these results suggest that degradation was likely not occurring and water would still be replenished in soil during vulnerable dry periods.

On pastures managed with near-continuous grazing of bermudagrass/rye (*Secale cereale* L.) in Georgia, soil penetration resistance was slightly greater (1.5 MPa) when grazed than not grazed by cattle (1.1 MPa), but this may have been partly due to the indirect effect of greater soil water content ($134\ g\ kg^{-1}$ without cattle and $120\ g\ kg^{-1}$ with cattle)[20]. There was also no difference in steady-state water infiltration between these two pasture systems, similar to our results under dry soil conditions.

On a Typic Cryaquept with >70% clay in Finland, steady-state water infiltration was greater in 3.5-year-old pasture (*Phleum pretense/Dactylis glomerata*) in areas with no visible trampling ($1.2\ mm\ min^{-1}$) than in areas with some trampling ($0.5\ mm\ min^{-1}$), with signs of penetrated hooves ($0.3\ mm\ min^{-1}$), and at drinking sites with destroyed vegetation ($0.2\ mm\ min^{-1}$)[21]. On a nearby coarser-textured soil (6% clay), steady-state water infiltration was also greater in areas with no visible trampling ($2.2\ mm\ min^{-1}$) than near drinking sites with destroyed vegetation ($0.4\ mm\ min^{-1}$). Deep hoofprints associated with poached pasture near drinking sites are not usually

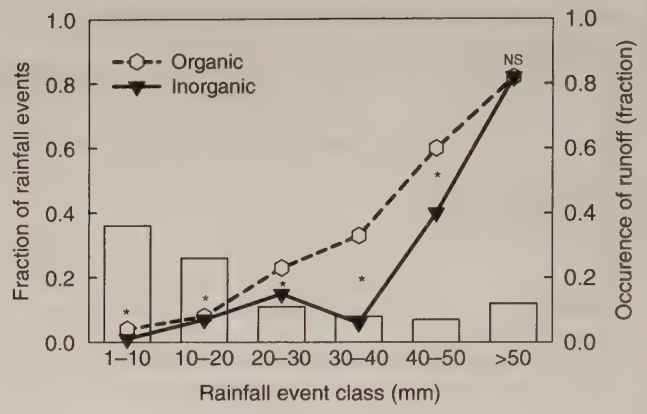

Figure 3. Frequency distribution of rainfall events (bars, left axis) and occurrence of runoff (lines, right axis) from December 1999 to December 2004 at the grassland site near Farmington, GA. Asterisk indicates significant difference ($P \leq 0.05$) in occurrence of runoff between nutrient-source treatments within a rainfall event class. NS indicates no significant difference.

produced immediately upon treading wet soil, but only after a progressive loss of soil strength due to repeated treading[22]. We did not measure infiltration near heavy-use areas, but rather only in the main grazing portion of paddocks to assess generalized effects of cattle on the majority of pasture land area. However, reduced infiltration in heavy-use areas in our study may have also occurred.

Rainfall and occurrence of runoff

Nearly 40% of the rainfall events during years 7–11 of this long-term study were small, ≤10 mm (Fig. 3). Seventy percent of rainfall events were ≤20 mm. Precipitation during these 5 years was 89 ± 26% of the long-term mean of 1250 mm. Precipitation during year 7 was the lowest at 587 mm and during year 10 was highest at 1431 mm.

Occurrence of runoff varied strongly with the size of a rainfall event (Fig. 3). With the majority of rainfall events ≤20 mm, there was ≤10% occurrence of runoff. When rainfall events were 20–40 mm, occurrence of runoff rose to 20%. Occurrence of runoff was 50% with rainfall events of 40–50 mm and was 82% with rainfall events >50 mm. Interestingly, there was significantly greater occurrence of runoff with organic only than with inorganic only as nutrient source in rainfall events that were < 50 mm in size. The reason for the difference in runoff occurrence with size of rainfall event was obviously due to mass flow of water that could not penetrate soil in a timely manner, but the reason for the difference due to nutrient source was not readily apparent. One possibility is that surface amendment with broiler litter may have increased hydrophobicity of the soil surface (in addition to surface residues), leading to greater incidence of runoff with smaller rainfall events. This possibility deserves greater attention in a more specifically designed research experiment. We did not anticipate this effect in our study, and therefore, were not prepared to address this issue further.

These data suggest that only a small fraction (15%) of rainfall events produced widespread occurrence of runoff (>50% of collectors) under pasture conditions with high surface residue and soil organic C content (i.e., low grazing pressure). This observation is consistent with observations from long-term watershed studies, in which the majority of sediment loss in conservation-managed agroecosystems is often associated with a few extremely large precipitation events[23,24]. However, these data should not be interpreted as a quantitative measure of runoff volume, since the observations were only of occurrence, whether that occurrence was large or small. Therefore, contrary to agricultural management systems with poor surface condition (e.g., degraded pastures or clean-tilled cropland fields subject to surface sealing), the relatively low occurrence of water runoff suggests that these pastures had infiltration capacities that could handle most rainfall events. Surface sealing is a function of iron oxide and soil organic C concentrations[25]. Kanhapludults in Georgia are prone to surface sealing when exposed to rainfall impact and lack of vegetation cover[26].

Relationships between variables

A total of 36 observations (3 nutrient source × 4 harvest management × 3 replications) were tested for correlation among several response variables (water infiltration, macropore filling, penetration resistance, soil water content, bulk density, soil organic C, and surface residue C and N). Only macropore filling, either in the wet period of March or in the dry period of April, was not related to any other variable. The rate of water infiltration during the wet period in March was negatively related ($P \leqslant 0.01$) to soil water content ($r = -0.57$), penetration resistance at 0–10 cm depth ($r = -0.50$), bulk density at 3–6 cm depth ($r = -0.53$) and bulk density at 0–12 cm depth ($r = -0.42$). The rate of water infiltration in March was also positively related to surface residue C ($r = 0.47$) and N ($r = 0.48$), and to soil organic C at 12–20 cm depth ($r = 0.42$). The rate of water infiltration during the dry period in April was only related to bulk density at 0–12 cm depth ($r = -0.42$). The rate of water infiltration between March and April, although different in magnitude due to major difference in soil water content, was positively related ($r = 0.55$).

Penetration resistance at 0–10 cm depth was negatively related to surface residue C ($r = -0.56$) and N ($r = -0.51$), and bulk density at 0–3 cm depth ($r = -0.49$), but positively related to bulk density at 3–6 cm depth ($r = 0.45$). Penetration resistance at 10–20 and 20–30 cm depths was not related to any other soil and water variable, but was highly related to resistance values at different depth increments. Interestingly, penetration resistance at 0–10 cm depth was positively related to soil organic C concentration at 0–3 cm depth ($r = 0.61$) and soil organic C content at 0–6 cm depth ($r = 0.51$). We expected a negative relationship between penetration resistance and soil organic C content, but our results suggest that accumulation of soil organic C near the soil surface may be due to high density and penetration resistance below 6 cm depth. Soil organic C was greater in grazed than in ungrazed management systems (Table 2). Although bulk density below 6-cm depth was not negatively affected by grazing[16], arrangement of pores and altered soil physical structure still may have sufficiently influenced penetration resistance between grazed and ungrazed systems. When data were separated into grazed and ungrazed treatments, there were weak, but negative relationships between penetration resistance and soil organic C at various depths in both systems. Further research on defining the relationship between soil organic C and penetration resistance within small depth increments in the soil profile is warranted to better understand how forage management systems might alter soil-surface structural conditions with time.

Correlations between response variables helped to strengthen the reasons for soil structural changes that occurred in this soil as a result of long-term pasture management treatments. For example, the reduction in water infiltration with increasing cattle grazing pressure was due to compression of soil that led to greater penetration resistance in the surface at 10 cm depth and greater bulk density, particularly at 3–6 cm depth. Greater surface residue accumulation with grazed and unharvested treatments than with haying led to greater soil organic C (Table 2), both of which likely assisted with the development of biopores for water infiltration to remain high, even in animal-trafficked soil with grazing. The buffering effect of surface residue and high surface-soil organic matter on the compressive force exerted from frequent cattle traffic in grazed pastures appeared to be significantly strong enough to avoid surface sealing and restriction of water intake. These results are consistent with observations reported by Watts and Dexter[27], in which integrity of soil aggregates from samples with high soil organic C (permanent grass) was maintained with mechanical stress, as compared with disintegration of soil aggregates from samples with low soil organic C (arable and clean fallow). In accordance, both water-stable macro-aggregation (>0.25 mm, $\sim 0.7\ g\ g^{-1}$) and macro-aggregate stability ($\sim 0.9\ g_{wet}\ g^{-1}{}_{dry}$) were high at the end of 4 years in our study[28].

Our results on grazing impacts on surface-soil structural conditions in bermudagrass/tall fescue pastures in Georgia have some similarities to a study of continuous versus rotational grazing of tallgrass praire in Texas[29]. In both studies, infiltration was greatest and penetration resistance was lowest in ungrazed exclosures. In both studies, bulk density was unaffected by the management system. We did not find differences in soil water content, macropore filling, infiltration rate, penetration resistance or soil organic C between low and high grazing pressures, but Teague et al.[29] observed: (a) lower soil water content in heavy-continuous grazing compared with light-continuous and multi-paddock grazing; (b) no difference between treatments in infiltration rate; (c) greater penetration resistance in heavy-continuous grazing compared with light-continuous and multi-paddock

grazing; and (d) lower soil organic C in heavy-continuous grazing than light-continuous and multi-paddock grazing. There is a great deal of interest in management-intensive rotational grazing[30–32], but there are very few studies documenting the quantitative impacts of such systems on surface-soil structural conditions, and therefore, such research is greatly needed to understand its impacts on contemporary environmental issues of soil and water quality, greenhouse gas emissions, and ecosystem structure and function.

Conclusions

A rather complex arrangement of surface-soil structural conditions developed in this Typic Kanhapludult in response to nutrient source and harvest management. The application of organic nutrients such as broiler litter had little impact on ponded water infiltration and penetration resistance, suggesting that how forage utilized applied nutrients and how harvest mechanisms manipulated forage-residue placement and quality were more important than the source of nutrients, *per se*. Although water infiltration was lower with grazing and haying than unharvested forage, as expected, this effect was significant only under wetter soil conditions. In addition, soil penetration resistance was greater with increasing grazing pressure. Lack of infiltration differences between grazed and ungrazed systems when soil was dry, suggests that pore connectivity may have been positively influenced by greater soil organic C in grazed systems, and this consequence may have compensated for the negative influence of greater penetration resistance with animal traffic. Water infiltration in continuously grazed bermudagrass/tall fescue pastures was adequately maintained with low or high grazing pressure compared to haying, despite a firmer soil surface. This effect may have been mediated by high surface residue cover, which prevented soil from sealing. Although not specifically tested here, management-intensive rotational grazing might be expected to also produce similar results between grazed and ungrazed management systems due to its reliance on periodical forage accumulation and subsequent surface residue accumulation. The put-and-take stocking system used in this study is a research tool that acknowledges residual forage mass as a keystone property of pastures, but that creates a continuous and more uniform grazing pressure throughout the year by adjusting stocking rate as necessary to maintain a target forage mass, not by adjusting return interval of a high stocking density. These results have important implications for a pedon-based understanding of hydrologic consequences from the 19 Mha of managed pastures in the warm, humid region of the southeastern USA; namely that well-managed pastures can promote efficient water cycling and control of nutrient runoff into receiving bodies of water. Further research is needed to understand the linkages between field- and watershed-scale hydrology in perennial pastures and their implications on water quality, especially under large, but infrequent storm events that cause the greatest occurrence of runoff.

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Profitability of organic and conventional soybean production under 'green payments' in carbon offset programs

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Abstract

Emphasis on reducing emissions from the greenhouse gases (GHG), carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) has increased in recent years in the USA, primarily for industry, transportation, energy and agricultural sectors. In this study, we utilized on-farm data collected by the USDA-National Agricultural Statistics Service (NASS) Agricultural Resource Management Survey (ARMS), secured under an agreement with the USDA-Economic Research Service (ERS) to analyze the profitability of organic and conventional soybean production, based on changes that 'green payments' in a cap-and-trade system would introduce in agricultural markets in the USA. In particular, the analysis focused on establishing whether organic producers would be better positioned to sequester carbon (C) and reap the benefits of the C-offset scheme compared to conventional producers, given the differences in costs, management practices and environmental benefits between organic and conventional production methods. We estimated several changes in profitability of soybean producers according to management practices, incentives for the generation of offset credits, and increase in energy input prices that a potential cap-and-trade system may introduce in future agricultural markets in the USA. Survey data suggested that even with lower yields, conventional producers could profit from converting to organic agriculture, given organic price premiums. In addition, taking into consideration both direct and indirect costs, average cost for conventional-till (CT) organic soybean production was approximately 9% lower than no-till (NT) conventional production. With a C market and payments for soil C sequestration through potential Clean Energy legislation, additional profit could be accrued by organic producers, because organic production would have 28% greater ton CO_2 eq. $acre^{-1}$ yr^{-1} sequestration than conventional NT. Thus, the environmental benefits from GHG reduction could incentivize increased conversion from conventional to organic production across the USA.

Key words: organic, carbon sequestration, survey, cost of production

Introduction

The American Clean Energy and Security Act, which was passed in the US House of Representatives in 2009, but failed to pass the US Senate, proposed the establishment of a cap-and-trade system to regulate greenhouse gas (GHG) emissions in the USA. The Act portended significant changes for agriculture in the USA, and research on the relationship between agriculture and GHG emissions has increased in the past year. The primary agricultural GHGs include carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). Although industry, transportation and energy sectors were the greatest source of GHG emissions in 2007, agricultural soil-management activities, including fertilizer applications, were responsible for 67% of US N_2O emissions[1]. While agriculture is considered an uncapped sector with respect to regulation of GHG emissions, as established in Title V Section 501 (b) in the House bill, in Title V Section 502 (a), the bill proposes that '...the Secretary shall establish a program governing the generation of offset credits from domestic agricultural and forestry resources.' Thus, farmers who reduce their emissions

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potentially could derive a profit by selling their emission credits.

A cap on the energy sector will most likely increase agricultural input prices, such as electricity, fuel and domestically produced fertilizers and pesticides[2]. This will be particularly important for conventional agricultural producers who rely on petroleum-based inputs. Organic agriculture, on the other hand, avoids the use of synthetic fertilizers and pesticides, and relies on the use of manure, legumes and other natural sources of nitrogen (N)[3], thus eliminating synthetic N use and possibly reducing N_2O emissions[4]. Organic agriculture also has a lower energy requirement[5–7]. Moreover, there is an indication that the benefit of lower energy use in organic agriculture is accompanied by greater carbon (C) sequestration rates than in conventional agriculture[8]. Thus, organic producers may be better positioned to sequester C and reap the benefits of the C-offset scheme than conventional producers. Greene et al.[9] point out that the mitigation of environmental externalities with organic production systems could improve the environmental performance of US agriculture if the adoption rate of organic practices were higher. In this study, we analyzed the potential for GHG offsets, particularly from reduction in CO_2 and N_2O emissions, of organic and conventional soybean producers in the USA using on-farm survey data obtained through the USDA-National Agricultural Statistics Service (NASS) Agricultural Resource Management Survey (ARMS) in 2006. More specifically, we determined the economic value to individual soybean farmers from the environmental benefits associated with their production practices in a potential C-offset market.

GHG mitigation in agriculture

As defined by Lal[10], 'soil C sequestration' is plant removal via photosynthesis of the GHG, CO_2, with subsequent storage of fixed C as soil organic matter. Both organic and conventional cropping systems have the potential to induce C sequestration, but each system has different energy requirements, cost structures and relative intensities of their sources of GHG emissions. Consequently, each of these components affects farmers' profit functions in a different manner. Research analyzing the differential profit between organic and conventional farming systems based on potential for C sequestration, GHG mitigation and economic value in a C-based market has been limited. Information regarding the performance of organic and conventional farming practices based on varying management regimes, however, has increased in recent years.

Tillage effects. The main component of agricultural soil-management activities is tillage. Tillage practices can be classified according to the percentage of residue cover remaining on the soil after planting. Conventional-till (CT) leaves less than 15% of residue cover on the soil after planting due to its reliance on the use of intensive tillage equipment[11]. Reduced-tillage refers to the case where 15–30% residue cover remains after tillage in order to reduce water or wind soil erosion[12]; mulch-till, ridge-till and no-till (NT) leave 30% or more residue coverage and are referred to as conservation tillage practices. CT involves plowing or disking the soil to prepare for planting and also includes cultivation for weed management. CT has been criticized for depleting soil C[13], while reduced tillage has been proposed as a method to restore soil organic carbon (SOC), reduce the amount of C released into the atmosphere and maintain a higher level of C storage. The United Nations Environmental Program (UNEP) and the World Meteorological Organization (WMO) Intergovernmental Panel on Climate Change (IPCC) guidelines[14] identify tillage as one of the main management practices that affect soil C stocks in croplands. Furthermore, tillage practices also influence, among others, chemical pesticide and fertilizer use. West and Marland[15] (p. 439–440) suggested that 'optimizing agricultural management through efficient use of fertilizers, irrigation and tillage operations can often enhance C sequestration while simultaneously reducing the C emissions associated with agricultural inputs such as fertilizers and on-farm fuels.' Reduced tillage systems, on the other hand, may be subject to herbicide resistance, based on reliance on herbicides and genetically modified seeds to manage weeds. Some authors also disagree with the argument that tillage practices can alter C emissions from soils and contribute to GHG emission reduction. For example, NT might reduce direct energy consumption, but would increase indirect energy consumption through an increased use of inputs/herbicides compared to CT[3]. Moreover, Lal[10] discussed the importance of identifying those 'hidden' C costs related to the production of nitrogenous fertilizers and pesticides when evaluating a practice's potential for SOC sequestration. Organic agriculture relies on the use of tillage because, by definition, it precludes the use of herbicides. Research has shown that despite relying on intensive and frequent cultivation, organic systems can store more soil C than conventional systems[16]. Thus, determining which combination of farming and tillage practices is best for sequestering SOC and GHG mitigation has been the subject of much discussion. West and Marland[15] stated that conversion from CT to NT in soybean production would result in a net removal differential of $-371\,\mathrm{kg\,C\,ha^{-1}\,yr^{-1}}$. Along the same lines, West and Post[17] proposed using a factor of 16% increase in C sequestration when switching from CT to NT. Robertson et al.[18] found that C storage was three times greater in an NT system than an experimental organic system. It should be noted, however, that their results were based on samples taken only from the top 7.5 cm of soil and their organic system was not as diversified as is required for certified organic production. Other research corroborated NT C sequestration benefits primarily in the upper soil layer, while plow tillage (PT) systems sequestered equivalent C when the entire soil profile was examined[19,20]. When specifically evaluating

NT against organic agriculture systems, Teasdale et al.[21] found greater soil C and N concentrations at all depth intervals up to 30 cm in an organic system than in NT at the end of 9 years. In a follow-up trial, the authors found that corn grain yield was 18% greater under organic than NT. Results from a 10-year-old long-term experiment in Iowa demonstrated that a four-crop rotational organic system could sequester 0.53 Mg C ha^{-1} yr^{-1} in the top 15 cm[22], similar to converting from PT to NT[17].

N and liming impacts. Although N_2O is produced naturally in soils through nitrification and denitrification processes, fertilizer applications can increase the quantity of N in the soil, fostering further nitrification and denitrification, and, in turn, increasing N_2O emissions[1]. According to IPCC Guidelines for National Greenhouse Gas Inventories[23], both synthetic N fertilizers and organic amendments (e.g., manure) contribute to GHGs. Reducing N applications through monitoring tools, such as the pre-side-dress N test[24], can help ameliorate GHG emissions. Lime applications to adjust soil pH and improve plant growth also can affect GHG emissions. As indicated in IPCC guidelines[14], liming can lead to CO_2 emissions as the lime dissolves. Few references are available comparing organic and conventional systems in relation to liming practices, as both groups apply lime according to site-specific soil conditions.

Energy use in organic versus conventional systems

Calculations for energy use in agriculture should incorporate not only direct energy consumption but also indirect energy consumption derived mainly from synthetic inputs such as fertilizers and pesticides[25,26]. Pesticide manufacturing, based on crude petroleum and natural gas products[27], requires up to five times greater energy per kilogram than N fertilizer[28]. Energy inefficiency of conventional agriculture, relative to organic practices, is driven by manufacturing and distributing synthetic pesticides and fertilizers[3], which accounts for 70–90% of the costs of production[24]. Pimentel[6] found that organically grown corn and soybeans use 31 and 17% less fossil-energy inputs, respectively, compared to conventional corn and soybeans. Mäder et al.[5] found that organic systems required 34–53% less energy than conventional counterparts. Research in the UK[29] found that organic crops require 50% less energy input per unit of area than conventional crops, as a result of avoiding synthetic fertilizers and pesticides.

C-sequestration evaluation

Atmospheric CO_2 sequestration through agricultural soil management has been suggested as a way to mitigate GHG emissions[3,10,27,30,31]. The IPCC recommended that analyses of GHG mitigation be performed using CO_2 equivalent emissions (CO_2 eq. or CO_2e) in order to allow comparisons of all gas emissions. All non-CO_2 gasses, such as N_2O, are converted into CO_2 eq. using their global warming potentials (GWP), which are calculated over a 100-year period[23], with CO_2 assigned a value of 1, CH_4 a value of 25 and N_2O a value of 298 CO_2 eq. Despite the wide use of the IPCC's Emission Factor Database (EFDB), these models do not include specific values for organic farming; thus, as an estimate, conventional farming values are employed[32].

Organic systems have been reported to sequester more C than conventional systems[22]. The UK Soil Association[33] reviewed studies from 39 sites around the world and found, in 32 of these sites, soil C levels were 8–325% greater in organic systems. Pimentel et al.[34] reported an annual soil C increase of 981, 574 and 293 kg C ha^{-1} yr^{-1} in 22-year-old experimental organic animal-based, organic legume-based and conventional systems, respectively. Niggli et al.[35] estimated a potential sequestration rate of 500 kg C ha^{-1} yr^{-1}, while Marriott and Wander[16] reported that organic farming increased SOC concentrations by 14% relative to conventional agriculture. Similarly, Liebig and Doran[36] found that there was 22% greater SOC in the surface 30.5 cm on organic farms than on conventional farms.

C sequestration versus reduction in emissions

As discussed by Lewandrowski et al.[30], a controversial issue that may arise in the regulation of the proposed C-trading market is the idea that C-emission reduction and C sequestration are not equivalent, attributing a more permanent nature to GHG mitigation benefits. This concept is based on the fact that while emissions can be reduced in one case (often only to be re-introduced through polluting practices at a later period), C storage can be completely reversed after plowing or tillage, releasing sequestered C into the atmosphere. Potential solutions to the problem of emission reduction versus C sequestration include using trading ratios by which the credits given for C sequestration are less than what is being sequestered, or requiring a 20-year contract for sequestration activities.

Based on the analysis of USDA-NASS ARMS data of organic and conventional soybean production in 12 US soybean-producing states collected in 2006, we hypothesized that organic soybean producers would be better positioned than conventional producers to take advantage of 'green payments' from C-offset programs, assuming that the policy adopted to address non-permanence concerns would include C reductions and C sequestration as equivalent metrics. In order to analyze potential changes in the profitability of organic and conventional soybean production based on organic price premiums and the sale of emission credits, we evaluated profits of representative farmers under different scenarios, typical of Midwestern US conditions.

Materials and Methods

The USDA-NASS ARMS on soybean production

Permission to utilize the USDA-NASS ARMS dataset was obtained through a memorandum of understanding with

USDA-Economic Research Service (ERS). All data were coded by USDA and remained confidential throughout the analysis. The ARMS data included information about a producer's expenses, costs and production practices from 237 organic soybean producers and 1425 conventional producers in Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, South Dakota and Wisconsin. Data collected in the ARMS survey corresponded to one randomly selected field of each of the producers in the sample; data were expanded to the farm level by taking into account the number of acres on the selected farm. As McBride and Greene[37] determined, through the use of farm survey weights that are proportional to their probability of selection, the ARMS sample was expanded to represent soybean acreage in each state, and in this particular dataset, such weights were also useful to ensure that surveyed organic producers represented the correct proportion of the population, even though their share on this sample was purposely over represented. The number of acres that the weighted sample of the ARMS survey represented was 63,846,000, accounting for 85% of the 75,522,000 total US soybean acres in 2006[38]. The ARMS data included a sub-sample of organic producers that were identified through USDA-accredited state and private organic-certifying agencies. According to ERS data, the number of acres planted to organic soybean production in 2006 was 114,581; the weighted data in the ARMS survey accounted for 82,661 acres or approximately 72% of 2006 US organic soybean acreage[39].

Scenario evaluation

In our evaluation of profits of surveyed farmers under different scenarios, the first scenario represented the profit attained by a conventional producer who does not alter production practices despite incentives that the Clean Energy Bill would propose to reduce emissions and sequester C. The second scenario took into account a producer's decision of using NT to make a profit from selling emission credits, since conventional producers practicing NT would be reducing their direct energy-related costs (and emissions) and availing themselves of the bill's incentives. The third scenario considered the case of a farmer who produces soybeans using organic practices. This latter scenario took into account the differential profit from selling emission credits and also the organic price premiums that a producer would obtain in the market. The equation that nested all the above-described scenarios is

$$\Pi_{\$/\text{acre}} = P_{\text{soy}} * Y_{\text{soy}} - C_{\text{soy}} + P_{\text{permit}} * Q_{\text{permit}} \quad (1)$$

where P_{soy} is the organic or conventional market price of a bushel of soybeans, Y_{soy} is the farmer's soybean yield in bushels per acre; C_{soy} is the per acre cost associated with the production method chosen and based on the ARMS soybean data. The last two terms represent the price and quantity (on a per-acre basis) of permits a farmer would be able to sell depending on practices chosen. An important aspect of the analysis of a representative producers' profit function is that the adoption of each practice might imply not only different input uses or C-sequestration rates but also different levels of output and sale price. This implies that while profit in Equation 1 could be higher due to lower energy use or higher C-sequestration rates using a specific agricultural practice, that practice could also convey lower yields, causing a simultaneous profit reduction. Thus, we calculated the profitability of a representative producer's choice taking into account all effects that the adoption of each alternative endorsed.

Estimations for terms

We estimated the first two terms of Equation 1 using ARMS data. To estimate the two components of the last term of that equation, we used current relevant literature.

Prices, yields and costs. We estimated the costs of organic and conventional soybean production following McBride and Greene's[37] dividing of costs into operating costs, operating and capital costs, and total economic costs. However, we computed costs taking into account producers' tillage practices. In this way, we are able to analyze difference in costs producers may have had even within the same method of production, and, thus computed Equation 1 in a more precise manner. As an example, we compared fertilizer cost differentials between conventional producers using CT and those using NT. However, because surveyed producers were not asked to indicate their tillage practices directly, classification according to tillage practices was an estimate, based on previous planted crop and machinery used.

GHG emissions. To estimate N_2O emissions generated by US soybean producers from the use of N fertilizers, we used the procedure and equations defined as Tier 1 in Chapter 11 of Volume 4 of the IPCC guidelines[14]. As explained by EPA[1] (p. 6–24): 'the Tier 1 approach is based on multiplying activity data on different N inputs [...] by the appropriate default IPCC emission factors to estimate N_2O emissions on an input-by-input basis'. We used N application estimates to compute emissions derived from soil management activities and emission allowances corresponding to each of the different tillage practices reported. For that calculation, we followed equation (11.1) from IPCC[14] and used the default emission factor of 1% to estimate direct N_2O emissions from managed soils; we then transformed the N_2O–N emissions into N_2O and further converted them into CO_2 eq. using their GWP. Following the EPA's glossary, we used 'CO_2eq.' for GWP-weighted emissions (http://www.epa.gov/climatechange/glossary.html#C).

As indicated in Volume 4, Chapter 11 of IPCC guidelines[14], adding lime leads to CO_2 emission as the lime dissolves, and emissions from liming were given an emission factor of 0.12 or 0.13 depending on whether limestone or dolomite was used.

C-sequestration estimates. Because ARMS data were limited to a single year and did not include information

Table 1. Profitability and production costs of organic and conventional soybean producers by tillage practices.

	Type of farm					
	Conventional				Organic	
	NT	Mulch	Reduced	CT	Reduced[1]	CT
	Dollars per planted acre					
Gross value of production	259.00	259.62	254.86	250.81	272.32	463.99
Operating costs						
Seed	33.21	30.16	32.40	32.05	30.55	30.40
Fertilizer	13.29	11.19	12.64	11.23	10.35	9.59
Chemicals	15.64	12.81	12.42	12.12	0.00	0.02
Custom operations	6.11	5.27	6.23	7.66	7.05	5.68
Fuel, lubrication and electricity	8.90	14.00	15.47	17.65	14.97	24.95
Repairs	9.44	12.28	13.47	13.64	11.72	17.88
Purchased irrigation water	0.00	0.08	0.13	0.53	0.00	0.05
Hired labor	0.80	1.44	1.35	2.00	1.65	13.40
Operating capital	2.06	2.04	2.21	2.26	1.77	2.11
Capital ownership costs						
Capital recovery	54.95	63.77	65.55	62.67	60.17	77.29
Taxes and insurance	7.62	7.71	8.81	10.49	9.57	14.43
Other costs						
Opportunity cost of unpaid labor	12.29	17.17	19.38	21.03	25.89	42.33
Opportunity cost of land	87.66	91.06	85.93	77.74	71.33	79.87
General farm overhead	13.09	12.38	14.08	15.93	26.37	25.88
Cost summary						
Operating costs	89.45	89.27	96.33	99.14	78.07	104.08
Operating and capital ownership costs	152.02	160.75	170.69	172.30	147.81	195.79
Total economic costs	265.06	281.35	290.09	287.01	271.40	343.87
Value of production less						
Operating costs	169.54	170.36	158.53	151.66	194.25	359.91
Operating and capital ownership costs	106.98	98.88	84.17	78.50	124.51	268.19
Total economic costs	−6.07	−21.73	−35.23	−36.20	0.92	120.12
Yield (bushels per planted acre)	46.83	47.73	46.83	45.86	19.58	31.64
Price (dollars per bushel)	5.52	5.44	5.45	5.46	13.57	14.32
No. of observations	633	408	213	166	20	217

[1] This category includes 18 producers who used reduced-till and two producers who used mulch-till. To avoid compromising confidential information, they are grouped together under this category in all tables.

regarding soil C stock in each field, we estimated C-sequestration rates available from current scientific literature. As Lal et al.[40] indicated, adopting conservation tillage simultaneously increases C sequestration and decreases GHG emissions. No specific data were available on CO_2 eq. reduction from organic systems, so conventional rates were used. According to the US EPA[41], conversion from conventional to reduced tillage could sequester 0.73 t CO_2 eq. $acre^{-1}$ yr^{-1}, a value within the range of 0.64 and 1.05 t CO_2 eq. $acre^{-1}$ yr^{-1}, cited by the Congressional Budget Office[42].

Results and Discussion

Conventional and organic soybean costs of production, yields and sale price

Conventional soybean production from the ARMS weighted sample was 28,544,000 acres of NT, 380,000 acres of ridge-till, 19,136,000 acres of mulch-till, 9,652,000 acres of reduced-till and 6,051,000 acres of CT. Organic soybean production included 4000 acres of reduced-till and 79,000 acres of CT. The organic ridge-till sample had only five observations and was removed from the descriptive and comparative analysis. According to survey data, there were significant differences in operating costs between organic and conventional farming (Table 1), and between different tillage practices in conventional farming, as evidenced by differences in chemicals, fuel, lubrication, electricity, repairs and hired labor costs. As one would expect, NT farms had the highest costs for chemicals, and those costs decreased as tillage intensity increased. Fuel, lubrication and electricity costs were greatest for organic producers who used CT. Thus, conventional NT farmers reported the lowest fuel, lubrication and electricity costs, confirming a degree of substitution between costs for chemicals and tillage operations. Repairs had a similar pattern. A greater

use of chemicals also reduced costs for hired labor and opportunity cost of labor. Thus, the data corresponded with the assertion by Pautsch et al.[43] that higher fuel, labor and machinery costs accrued for CT than conservation tillage practices. Nevertheless, soybean is among the crops with the lowest energy-input costs as a share of total operating costs, accounting for approximately 22% of total operating costs in 2004[24]. In 2006, fertilizer, fuel, lubrication and electricity accounted for 27% of total operating costs[37].

NT had the lowest total economic costs. Organic producers who used CT had the highest total economic cost (without considering ridge-till). Organic producers who used reduced-till and not chemical herbicides reported significantly lower labor costs than organic farmers using CT. Moreover, the former group had labor costs that were lower than conventional producers who used CT. Furthermore, organic producers using reduced-till had the lowest operating costs; however, it also evident that they were compromising yields in the process. It is not clear from the data that reduced labor costs were strictly the result of a reduced number of field operations (i.e., tillage) or less 'walking' of weeds in organic reduced-till fields. However, the number of observations in the reduced-till sample was low, casting doubt on the true representation of these results.

Organic producers using CT obtained soybean yields that were approximately 70% of their conventional counterparts (Table 1), similar to results reported by McBride and Greene[37], but different from other reports. Badgley et al.[44] conducted the most comprehensive review of organic versus conventional yields worldwide and found few yield differences between the two production systems. Delate et al.[45] reported equivalent organic and conventional corn and soybean yields under Midwestern US conditions. Pimentel et al.[34] also confirmed similar yields over 22 years of a long-term comparison trial. Mäder et al.[5] obtained organic yields that were between 80 and 100% of conventional yields for all crops over 21 years in an organic rotation of wheat, potato and grass–clover hay. Other studies reported corn yield in an organic system reaching 92% of conventional corn yield[46], taking into account yields during transition from conventional production. In the same study, organic soybean yield was the same as conventional soybean yield. Porter et al.[47] reported organic corn yields 7–9% lower and organic soybean yields 16–19% lower than conventional crop yields. Recently, Singerman et al.[48] surveyed organic corn and soybean producers in three major organic soybean-producing states and also found similar results to the ARMS data. The dissimilarity between these survey data and the literature could be due to weeds being more easily managed in experimental plots than on private farms. In addition, the genetic background of the different soybean types/varieties planted in organic versus conventional farms may have impacted results: 80% of ARMS-surveyed organic farmers planted the larger-seeded, high-protein, tofu-type soybeans for the food-grade market, which traditionally have had lower yields than the smaller-seeded, herbicide-resistant feed-grade soybeans that 96% of conventional producers reported growing.

Despite lower yields and higher costs reported by organic producers, CT organic farmers had positive economic results, with an average profit of approximately $120 per planted acre (Table 1). Conventional soybean producers, on the other hand, did not derive a profit, based on their costs of production, yields and sale price, with the NT group achieving the greatest returns at a loss of approximately $6 per planted acre. Profits from organic soybean production were a consequence of the price premiums that organic producers were able to obtain in the market, which more than offset lower yields. These results correspond with those reported by Delate et al.[45,49] in which profit from organic soybean production was significantly greater than from conventional production. Singerman et al.[50] reported that organic soybean prices from 2004 to 2009 were consistently and significantly higher than conventional soybean prices with approximately the same risk as conventional production. Thus, even without the C-offset market, organic price premiums could incentivize conventional soybean producers to convert to organic practices to increase their profits. However, as Clarkson[51] pointed out, the 2-year period in which transitioning producers would not obtain organic price premiums; a perceived increased production risk with organic production; the different marketing structure compared to delivering the crop to the local elevator; and less governmental support, in terms of adequate crop insurance, research and extension support, discourage conventional farmers from considering organic production.

Although 'crop rotation' was not enumerated in the ARMS dataset, we can speculate on the effect of different crop rotations on profitability, based on data derived from Midwestern US organic systems for that period[52]. In 2006, returns to management for organic soybeans in a corn–soybean–oat–alfalfa rotation were $231 acre^{-1} with a rotational average of $180 acre^{-1}. This value compares to $67 that was returned to management for the conventional corn–soybean rotation, a difference of $113 acre^{-1}. For the 2006–2010 time period, the average difference was $211—again favoring the organic rotation.

Fertilizer use and GHG emissions estimates

Estimates of quantities of commercial fertilizers, other organic N additions (e.g., manure), lime and active ingredients of chemicals that surveyed producers used, according to tillage practices on a per-acre basis, were less on organic than conventional farms in all categories except for manure (Table 2). According to the weighted sample, conventional soybean producers used 161,372,000 lb N for soybean production in 2006, compared to 16,532 lb N in organic systems. Using computations for N_2O emissions converted into CO_2 eq. (Table 3), organic producers using CT generated 9 and 33% less CO_2 eq. emissions from N additions than conventional producers using NT and CT,

Table 2. Commercial and organic fertilizer, soil conditioner and chemicals quantities by tillage practice[1].

	Type of farm						
	Conventional				Organic		
	NT	Mulch	Reduced	CT	Units	Reduced	CT
	Quantity per acre						
Commercial fertilizer							
N	2.5	2.6	2.8	2.4	lb	0.0	0.2
Phosphorus	9.7	7.9	12.0	7.0	lb	0.2	0.4
Potassium	21.1	18.2	16.9	18.8	lb	0.5	0.4
Organic N additions							
Quantity of N in manure	0.6	1.7	0.9	1.8	lb	2.2	2.6
Quantity of phosphorus in manure	0.2	0.6	0.4	0.6	lb	0.7	1.0
Quantity of potassium in manure	0.6	1.9	1.0	1.9	lb	0.9	1.8
Soil conditioner							
Lime	0.26	0.16	0.19	0.18	ton	0.06	0.14
Chemicals active ingredient							
Herbicide	1.3	1.0	1.0	1.1	lb	0.0	0.0
Insecticide	0.20	0.60	0.70	0.50	oz	0.0	0.0
Fungicide	0.04	0.03	0.01	0.06	oz	0.0	0.0
No. of observations	633	408	213	166		20	217

[1] Calculations based on the assumption that the entire planted acreage of soybean was treated.

Table 3. CO_2 eq. emissions from N and lime additions by tillage practice (kg CO_2 eq. $acre^{-1}$ yr^{-1}).

	Type of farm					
	Conventional				Organic	
	NT	Mulch	Reduced	CT	Reduced	CT
N	5.3	5.4	6.0	5.1	0.0	0.4
Quantity of N in manure	1.2	3.6	2.0	3.7	4.6	5.5
Total N	6.5	9.0	8.0	8.8	4.6	5.9
Lime	0.11	0.07	0.08	0.08	0.03	0.06

respectively. Since no questions were asked of surveyed producers regarding the exact type of material used for liming, we assumed that all producers used limestone, a typical material used in the Midwestern US. CT organic producers generated 45 and 25% less CO_2 eq. emissions from lime than conventional producers using NT and CT, respectively (Table 3).

Energy use between organic and conventional production

Data in the ARMS survey were limited to farm-operation activities and costs; thus, we used pesticide and fertilizer consumption as a proxy for energy inputs since energy use in on-farm operations was indirectly contained in the dataset (e.g., through fuel, electricity, fertilizer and pesticide quantities). Although there are excellent estimators of farm energy use in tools such as the USDA-Natural Resources Conservation Service (NRCS) Energy Estimator[53], the NRCS calculator uses average estimates based on numerous sources, while we utilized ARMS data only in order to be consistent in our analysis. Using data from Table 1, CT organic soybean producers reported higher direct-energy consumption, as the costs of fuel, lubrication and electricity were 280 and 41% higher than from conventional producers using NT and CT, respectively. However, when one considers both direct and indirect energy use (i.e., including energy use costs of fertilizers and pesticides), average energy cost for CT organic producers was about 9% lower than that for NT conventional producers. Direct and indirect energy use costs for organic producers were $34.57 versus $37.83 per acre for conventional producers.

Additionally, there are energy costs that are unaccounted, but accrue during manufacture of petroleum-based inputs, but we only analyzed for an individual farmer's profit-maximizing decision. Thus, since the 'positive externality' derived from lower indirect energy consumption in organic soybean production was not internalized, we did not include the emission calculations. However, this calculation could be considered when taking into account the societal benefits and lower environmental footprint of organic agricultural practices.

As indicated in a preliminary analysis report, USDA-ERS[54] estimated that the Clean Energy Bill would increase the price of petroleum, electricity and natural gas by 3.2, 7.4 and 10.7%, respectively, by 2015. The report also indicated that even though direct energy prices would rise immediately, affecting the agricultural sector, the impact of the legislation on crop producers would be minimized, due to the Bill's provisions for energy-intensive, trade-exposed

Table 4. Fuel, lubrication and electricity costs for machinery and irrigation; and fuel cost of drying, by tillage practice.

	Type of farm					
	Conventional				Organic	
	NT	Mulch	Reduced	CT	Reduced	CT
	----------Dollars per planted acre----------					
Machinery	7.04	11.87	12.68	13.87	14.63	22.36
Irrigation	1.84	2.09	2.70	3.65	0.00	2.47
Fuel drying cost	0.03	0.04	0.09	0.13	0.35	0.12
Total	8.90	14.00	15.47	17.65	14.97	24.95

entities (EITE) to be included, such as N fertilizer companies. Thus, USDA-ERS concluded that due to the EITE provisions, greater costs primarily would be a consequence of fuel costs, with greater impact on fuel-intensive crops, such as rice and sorghum. With soybean capable of fixing its own N, soybean crop producers would be minimally impacted; UDSA-ERS estimated an average increase of $0.45 per acre from 2012 to 2018.

We used energy cost data to compare costs of fuel, lubrication and electricity, classified by machinery use, irrigation and drying, between organic and conventional soybean producers, which showed greater machinery costs in organic production (Table 4). We then estimated the impact of the Clean Energy Bill on energy inputs of soybean producers in our profitability analysis.

Changes in profitability derived from the C-offset market

We evaluated the impact of 'green payments' similar to those promulgated in the American Clean Energy and Security Act of 2009 on the profitability of soybean producers. In order to measure the differential emissions of CO_2 eq. from the two most relevant methods of production (conventional NT and CT organic), we used emissions generated by CT, conventional soybean production as the baseline. We used the value of 20% greater C sequestration with the adoption of organic practices from the Soil Association[33], as this was the most comprehensive review of C-sequestration comparisons between conventional and organic methods of production. The value was also very similar to that of Liebig and Doran[36] from surveyed organic farms, and was within the range reported by Rodale Institute[8] from their experiments.

Thus, based on the literature, as well as on EPA's estimate of additional CO_2 eq. sequestration from conversion to reduced tillage and information from Table 3, we constructed Table 5, which shows the differential emission reductions and sequestration rates between NT and CT organic production. Conventional NT and organic soybean production had a sequestration rate of 0.7 and 0.9 t CO_2 eq. acre^{-1} yr^{-1} compared to conventional production using

Table 5. Differential emission reductions and sequestration rates between NT and organic practices with respect to CT (t CO_2 eq. acre^{-1} yr^{-1}).

	NT	Organic CT
Tons of CO_2 eq. reduced from		
N	0.0015	0.0029
Liming	−0.03	0.02
Tons of CO_2 eq. sequestered from		
NT practices	0.73	–
Organic differential	–	0.88
Total	0.70	0.90

CT. Organic production was 28% greater in t CO_2 eq. acre^{-1} yr^{-1} sequestration than NT.

The Congressional Budget Office[55] estimated that the price of GHG allowances would be $16 per t CO_2 eq. in 2012 and would rise to $26 per t CO_2 eq. in 2019. EPA[41] estimated that the allowance price would be about $13 per t CO_2 eq. in 2015 and would rise to $16 by 2020. Table 6 shows the increased profitability of producers using NT and CT organic production with the introduction of C markets. Values represent sequestration rates (Table 5) multiplied by corresponding allowance price. Since we assumed that EITE provisions would prevent fertilizer and chemical prices from increasing until 2025, we computed no change in their prices until then. Other agricultural input prices would likely increase as the consequence of the rise in prices for petroleum, electricity and natural gas; therefore, we used USDA's[54] estimates to compute their effect on soybean producers' profitability. Thus, we multiplied the costs in Table 4 for machinery by the estimated increase in petroleum price for years 2012, 2020 (used as a proxy for 2019) and 2025, i.e., by 3.2, 4 and 4.7%, respectively. In the case of irrigation, we multiplied costs by the estimated increase in electricity prices for the same years, i.e., by 10.7, 12.7 and 14%, respectively. For drying costs, we multiplied costs by the estimated increase in natural gas prices, i.e., by 7.4, 8.5 and 8.6%, respectively, for each year. For the 2025 scenario, we multiplied fertilizer and chemical costs in Table 1 for each of the corresponding tillage practices by the estimated increase for natural gas in that year (8.6%).

In 2012 (2025), conventional producers using CT would lose an additional $0.84 ($3.18) per planted acre, whereas NT conventional producers and CT organic producers would benefit from the creation of a cap-and-trade system. NT producers would increase their profit by $8.65 ($15.07) per planted acre, while CT organic producers would increase profit by $10.67 ($21.08) per planted acre. Thus, adopting the economic results from Table 1 as a reference, *ceteris paribus*, in 2012 (2025), conventional producers that use CT will have an average total loss of $37.04 ($39.38) per planted acre; NT producers would have an average profit of $2.58 ($9) per planted acre; and CT organic producers would have an average profit of $130.79

Table 6. Comparison of changes in profits among CT, NT and CT organic, derived from the implementation of the C-offset markets.

	2012			2019				2025		
	$\$ t^{-1} CO_2$ eq.									
	13			16		26		26		
Allowance price	**Conventional CT**	**Conventional NT**	**Organic CT**	**Conventional NT**	**Organic CT**	**Conventional NT**	**Organic CT**	**Conventional CT**	**Conventional NT**	**Organic CT**
Additional profit from	Dollars per planted acre									
Sequestered t CO_2 eq. $acre^{-1} yr^{-1}$	0.00	9.08	11.66	11.17	14.35	18.15	23.32	0.00	18.15	23.32
Less expected increase in operating costs										
Fertilizer				EITE provision				0.97	1.14	0.83
Chemicals				EITE provision				1.04	1.34	0.00
Machinery	0.44	0.23	0.72	0.23	0.72	0.28	0.89	0.65	0.33	1.05
Irrigation	0.39	0.20	0.26	0.20	0.26	0.23	0.31	0.51	0.26	0.35
Drying	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.01	0.00	0.01
$\Delta\pi$	−0.84	8.65	10.67	10.75	13.36	17.63	22.10	−3.18	15.07	21.08
$\Delta\pi$ from CO_2 eq. sequestration for NT	−9.08	–	2.58	–	3.18	–	5.16	−18.15	–	5.16
Total $\Delta\pi$ for NT	−9.50	–	2.02	–	2.61	–	4.46	−18.26	–	6.01
$\Delta\pi$ from CO_2 eq. sequestration for organic	−11.66	−2.58	–	−3.18	–	−5.16	–	−23.32	−5.16	–
Total $\Delta\pi$ for organic	−11.51	−2.02	–	−2.61	–	−4.46	–	−24.26	−6.01	–

($141.20). Due to EPA's differential sequestration rate attributed to NT, profitability per acre would increase $9.08 ($18.15) when switching from CT to NT in 2012 (2025). Similarly, profitability per acre would increase $11.51 ($24.26) when switching from conventional agriculture using CT to organic agriculture. The total differential in profitability between NT and organic CT starts at $2.02 per acre in 2012 if the price of permits would be $13 t CO_2 eq.; the differential increases to $4.46 per acre in 2019 and to $6.01 per acre in 2025 when the EITE provisions would be scheduled to expire.

Conclusions

Using the USDA-NASS ARMS dataset for soybean production in 2006, we estimated changes in soybean production profitability based on surveyed producers' management practices, incentives for the generation of GHG offset credits, and the increase in energy input prices that a cap-and-trade system would introduce in agricultural markets in the USA. Conventional producers would profit from conversion to organic agriculture, especially after the EITE provisions expire in 2025. These conclusions are particularly pertinent for conventional soybean producers that use CT, not only because the differential profit between CT with herbicides and CT organic production is the greatest but also because this group would incur the greatest losses with the introduction of the C-offset market. These conventional producers could more easily convert to NT production and realize economic benefit. Finally, given the expected increase in energy prices over the next decade and the positive externality derived from lower indirect energy consumption, organic soybean production offers significant economic benefits compared with conventional production with or without NT.

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Effects of harvest period, nitrogen fertilization and mycorrhizal fungus inoculation on triticale (× *Triticosecale* Wittmack) forage yield and quality

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Abstract

The practice of grazing winter cereals before heading and subsequently harvesting the forage is common in some countries. Triticale (× *Triticosecale* Wittmack) is particularly interesting due to its nutritional characteristics, and forage yield and quality are strongly influenced by agronomic techniques. The effect of two modes of utilization (cut at heading stage and cut during winter grazing simulation with regrowth cut at heading stage), three nitrogen (N) fertilization levels [0 and 100 kg ha^{-1} split in two phases; and 100 kg ha^{-1} before sowing, using a fertilizer with 3,4-dimethylpyrazole phosphate (DMPP) as a nitrification inhibitor (NI)] and mycorrhizal fungus inoculation (mixed inoculation of *Glomus intraradices* and *Glomus moseae*) on forage yield and quality, protein fractions and *in vitro* digestibility of triticale was determined in southern Italy. Mode of utilization and fertilization affected significantly forage yield and quality, while mycorrhizal fungus inoculation influenced positively some quality parameters. Cutting at heading stage of triticale led to high dry matter (DM) production compared to the winter cut simulating grazing, but no differences in protein yield (PY). N fertilization increased total DM and PYs, but no differences were reported among the application modalities of N fertilizers. Triticale protein fractions and protein and fiber digestibility were improved by N fertilization. Mycorrhizal fungus inoculation positively influenced forage quality at heading stage, which resulted in more digestible fiber and protein. As a result, when triticale was grown under Mediterranean conditions, mode of utilization and N fertilization resulted in greater forage yield and quality, while mycorrhizal fungus inoculation influenced positively forage nutritional quality parameters.

Key words: triticale, fertilization, mycorrhization, protein fractions, digestibility

Introduction

Triticale (× *Triticosecale* Wittmack) is grown for grazing, fresh forage, silage and hay under a wide range of production conditions. Furthermore, cereal straw is a main animal feed source, and the use of triticale as straw is in continuous expansion, especially in Mediterranean and semi-arid countries[1]. The practice of grazing winter cereals before heading and subsequently harvesting the forage is common in some countries[2]. Little research has been done on the effect of grazing on the forage yield of winter cereals. Hennessy and Clements[3] recommend a single light grazing in early spring because more intensive grazing of winter cereals results in reduced subsequent growth. Data on the effect of forage removal (grazing or clipping) on grain yield of cereal crops are available[4], while data on the effect of grazing on forage production were not well defined[5].

Nitrogen (N) fertilization has a direct effect on the plant N uptake and consequently the N present as a protein in animal feed. Mineral N fertilizer is regarded as a significant contributor to water pollution by nitrates and to atmospheric pollution by nitrous oxides, but there is little likelihood that adequate food supplies can be maintained without fertilizers[6]. The development of cropping strategies that increase N use efficiency (NUE) could reduce unnecessary input costs to farmers and environmental impact of N losses, while maintaining crop

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yield. Splitting of N fertilizer application has been suggested as a strategy to improve NUE in winter cereal, on the assumption that the timing of application has a significant effect on the N uptake by the crop[7,8]. Low efficiency attributed to N fertilizer application in autumn has been observed in a large number of studies, and justifies N applications in spring[9,10], particularly in Mediterranean climates. A further possible means of increasing NUE is the addition of a nitrification inhibitor (NI) to the fertilizer, particularly in the case of large applications. Since the 1960s, the fertilizer industry has developed compounds that delay bacterial oxidation of NH_4^+ to NO_2^- (first step of nitrification) to diminish nitrate losses and to increase N-fertilizer efficiency[11]. In one study, Zerulla et al.[12] proposed (3,4-dimethylpyrazole phosphate, DMPP) as an NI, effective when added to granulated N fertilizers at a rate of 0.5–1.5 kg ha^{-1}, and able to diminish N_2O emissions to the atmosphere[13].

Since several species of bio-activators have been isolated from the rhizosphere, many experiments have been conducted to test bacteria and endo-mycorrhizal fungi as inoculants to enhance the performance of plants, but few trials have been conducted on triticale[14]. Little work has been performed on the management of triticale as forage, specially on dry matter yield (DMY) and quality in terms of crude protein (CP) fractions and nutrient digestibility. Therefore, the objective of this work was to assess the effects of the mode of utilization, N fertilization and mycorrhizal fungus inoculation on forage yield, protein fractions and digestibility of triticale.

Materials and Methods

Field trial

A trial was conducted in southern Italy at Gaudiano di Lavello—Potenza (41°06′N; 15°51′E; 180 m above sea level) on a sandy-clay soil, characterized as sub-alkaline, low in total N (0.77‰; the Kjeldahl method) and high in available P (83 mg l^{-1}; the Olsen method) and exchangeable P (482 mg l^{-1}; $BaCl_2$; the TEA method). The experimental site was characterized by a summer-dry climate with a total annual rainfall of 450 mm distributed from Autumn to Spring and a mean temperature of 16°C. During the experimental period, the total rainfall was 258 mm and temperatures did not show any significant variation from the average. The effects of 12 different treatments obtained by the factorial combination of three N fertilization levels (N0 = without N; N100 = 100 kg ha^{-1} of N, 23 kg ha^{-1} as diammonium phosphate 18-46-0 before sowing followed by 77 kg ha^{-1} as ammonium nitrate top-dressing in February (Zadoks' stage 22 at tillering;[15]); N100E = 100 kg ha^{-1} of N as Entec 25-15-0 containing 3,4 DMPP®, applied before sowing), two modalities of herbage utilization (H = cut at heading stage for hay production; GSH = cut during winter grazing simulation and regrowth cut at heading stage for hay production) and two treatments of inoculation with (M) and without (M0) commercial mycorrhiza (Mikendo—Gitenbiological® at a rate of 1.5 kg of inoculum per 100 kg seeds) were studied. The inoculum used was based on a mixture of spores (*Glomus intraradices* and *Glomus moseae*), with a density of 0.6 kg l^{-1}, in the form of wettable powder, containing 2000 spores of arbuscular mycorrhizal fungus per gram. This product was formulated to be applied to seeds, before sowing, so that the mycorrhizal propagules remained adhered to the seed and would then be able to germinate and to colonize the new roots. Other ingredients in the formulation included: dextrose, vermiculite and clay (in order to help the propagules to adhere to seeds). Since N100 and N100E received 60 kg ha^{-1} of P_2O_5, treatment N0 received the same amount of P_2O_5 as superphosphate before sowing in order to give uniform P fertilization. A split–split plot experimental design was used with three replicates and sub-sub-plot area of 18 m^2; the 'mycorrhizal fungus inoculation', 'N fertilization' and 'mode of utilization' effects were tested in the whole plot, sub-plot and sub-sub-plots, respectively. Soil preparation was performed in November 2007, plowing the soil to a depth of 30 cm. The previous crop was safflower (*Carthamus tinctorius* L.). Triticale cv. Rigel (spring variety) was sown on December 12, 2007 at a seeding rate of 250 kg ha^{-1} and a row spacing of 20 cm. Forage at winter cut was removed at Zadoks' stage 31 (first node detectable[15]). Forage yield was determined by harvesting the whole plot.

Table 1. Effect of N fertilization and mode of utilization on DMY and PY of winter cutting plus heading cutting of triticale forage.

	DMY			PY		
Treatment	**H**	**GSH**	**Mean**	**H**	**GSH**	**Mean**
	----------t ha^{-1} DM----------					
N0	6.6	5.0	5.8^b	0.40	0.38	0.39^b
N100	11.7	9.7	10.7^a	0.95	0.92	0.93^a
N100E	11.9	10.9	11.4^a	0.81	0.94	0.87^a
Mean	10.1^y	8.5^x	9.3	0.72	0.74	0.73

N0, without N fertilization; N100, 100 kg ha^{-1} of N, 23 kg ha^{-1} of N as diammonium phosphate 18-46-0 before sowing followed by 77 kg ha^{-1} of N as ammonium nitrate as top-dressing in February; N100E, 100 kg ha^{-1} of N as Entec 25-15-0; H, cut at heading stage for hay production; GSH, cut during winter grazing simulation and regrowth cut at heading stage for hay production.
Means within a row (a, b) and column (x, y) with different superscript letters differ significantly ($P < 0.05$).
Values are expressed as the mean of three determinations per treatment.

Chemical analysis

Samples of each treatment were ground in a hammer mill with a 1 mm screen and analyzed in duplicate for

Table 2. Effect of mycorrhization and N fertilization on chemical composition of winter cutting triticale forage.

	Treatment							Significance		
	M0			M						
Item	N0	N100	N100E	N0	N100	N100E	SEM	M	N	M×N
	----------$g\,kg^{-1}$----------									
DM	173	163	152	171	158	150	4.7	ns	*	ns
	----------$g\,kg^{-1}$ DM----------									
CP	178	228	191	201	235	221	4.2	ns	*	ns
Crude fat	29	33	31	30	33	31	0.3	ns	*	ns
Ash	99	102	104	100	101	101	2.2	ns	ns	ns
NDF	541	540	542	540	535	539	5.8	ns	ns	ns
ADF	357	350	359	348	341	349	4.1	*	ns	ns
ADL	30	25	20	32	28	25	0.2	ns	*	ns

N0, without N fertilization; N100, 100 $kg\,ha^{-1}$ of N, 23 $kg\,ha^{-1}$ of N as diammonium phosphate 18-46-0 before sowing followed by 77 $kg\,ha^{-1}$ of N as ammonium nitrate as top-dressing in February; N100E, 100 $kg\,ha^{-1}$ of N as Entec 25-15–0; M0, without mycorrhiza inoculation; M, with mycorrhiza inoculation.
*$P<0.05$; ns, not significant.
Values are expressed as means of three determinations per treatment.

dry matter (DM), ash, CP (Kjeldahl N×6.25) and crude fat (ether extract with previous hydrolysis) according to the procedures described by the Association of Official Analytical Chemists (AOAC)[16]. Neutral detergent fiber (NDF) and acid detergent fiber (ADF) were determined according to Van Soest et al.[17], and were corrected for residual acid insoluble ash. Acid detergent lignin (ADL) was determined by the method of Van Soest and Robertson[18].

Fractionation of CP

Fractionation of CP was carried out by the Cornell Net Carbohydrate and Protein System[19]. According to this system, CP was partitioned into three fractions: fraction A was non-protein N (NPN)×6.25; fraction B was true protein and fraction C was unavailable protein. Fraction B was further divided into three sub-fractions B_1, B_2 and B_3 of rapid, intermediate and slow rates of ruminal degradation, respectively. CP fractions A and B_1 were soluble in borate–phosphate buffer, CP fraction B_2 was insoluble in the buffer but soluble in neutral detergent solution, and fraction B_3 was insoluble in the buffer and in the neutral detergent, but soluble in acid detergent. CP fraction C was insoluble in acid detergent (acid detergent insoluble protein, ADIP); and contained protein associated with lignin, tannin–protein complexes, and Maillard products known to be highly resistant to microbial enzymes. Precipitated true protein, buffer insoluble protein, neutral-detergent insoluble protein (NDIP) and ADIP were analyzed as described by Licitra et al.[20]. Fraction A was calculated as the difference between the total CP and precipitated true protein determined by Kjeldahl analysis of the residue resulting after precipitation with tungstic acid followed by filtration. CP fraction B_1 was estimated as true protein minus buffer insoluble protein, fraction B_2 as buffer insoluble protein minus NDIP and fraction B_3 by subtracting the ADIP (fraction C) from the NDIP.

In vitro *digestibility determination*

In vitro digestibility of DM (DMd), organic matter (OMd), CP (CPd) and NDF (NDFd) were determined using the Daisy^{II} Ankom® incubator system. The complete unit consisted of four incubation vessels with a capacity of 2 liters each. Each vessel contained 1.6 liters of buffer solution, 400 ml of either rumen liquor or faecal liquor as the inoculum and 24 nylon bags. Collected plant samples were ground to pass through a 1 mm screen. Each of the samples was digested in duplicate for each source of inoculum. Nylon filter bags (Ankom F57, ANKOM Technology, Fairport, New York) were rinsed in acetone and allowed to air dry before drying at 100°C for 24 h, after which dry bag weight was recorded. For each treatment, 0.25 g of ground sample was added to nylon bags, which were then dried at 105°C for 24 h, after which dry sample plus bag weight was recorded. Duplicate nylon bags for each feed type were randomly allocated to one of the four digestion vessels and to one of the four inoculum treatments. The microbial inoculum was prepared from rumen liquor collected from four mature healthy sheep using an esophageal tube under mild vacuum. The sheep were fed a diet sufficient for their maintenance allowance[21], which contained a 50:50 chopped alfalfa hay and grain mixture. After collection, the rumen liquor was placed in an air-tight container and transported to the laboratory at 39°C. The inoculum from each animal was filtered through eight layers of gauze cloth, purged with CO_2, and kept in a prewarmed thermos until use (within approximately 20 min). The 400 ml of inoculum were introduced in each flask and placed into a Daisy^{II} system.

Table 3. Effect of mycorrhization, N fertilization and mode of utilization on chemical composition of triticale forage cut at heading stage.

	Treatment										Significance					
	M0				M											
	H		GSH		H		GSH									
Item	N0	N100	N100E	N0	N100	N100E	N0	N100	N100E	SEM	M	G	N	M×N	M×G	G×N
	---g kg^{-1} DM---															
DM	258	285	296	268	261	283	307	278	280	6.2	ns	ns	ns	ns	ns	ns
CP	60	76	65	60	74	64	63	85	71	1.5	*	ns	*	ns	ns	*
Crude fat	16	17	15	15	16	15	16	18	16	0.2	ns	ns	ns	ns	ns	ns
Ash	96	98	99	95	97	99	102	102	103	4.4	*	ns	ns	ns	ns	ns
NDF	582	575	567	611	609	601	526	524	523	5.5	*	*	ns	*	*	*
ADF	369	366	360	385	381	381	354	353	352	3.9	*	ns	ns	ns	ns	ns
ADL	45	38	41	48	35	42	51	40	43	0.2	ns	*	*	ns	ns	ns

N0, without N fertilization; N100, 100 kg ha^{-1} of N, 23 kg ha^{-1} of N as diammonium phosphate 18-46-0 before sowing followed by 77 kg ha^{-1} of N as ammonium nitrate as top-dressing in February; N100E, 100 kg ha^{-1} of N as Entec 25-15-0; M0, without mycorrhiza inoculation; M, with mycorrhiza inoculation; H, cut at heading stage for hay production; GSH, cut during winter grazing simulation and regrowth cut at heading stage for hay production.

*$P<0.05$; ns, not significant.

Values are expressed as means of three determinations per treatment.

Each digestion vessel contained 400 ml of inoculum and 1.6 liters of buffer solution (1:4, v/v). The buffer solution[22] consisted of 1.331 buffer A (KH_2PO_4, 10.0 g l^{-1}; $MgSO_4.7H_2O$, 0.5 g l^{-1}; NaCl, 0.5 g l^{-1}; $CaCl_2.2H_2O$, 0.1 g l^{-1}; and urea, 0.5 g l^{-1}) and 266 ml of buffer B (Na_2CO_3, 15.0 g l^{-1} and $Na_2S.7H_2O$, 1.0 g l^{-1}), mixed in each digestion vessel and the pH was adjusted to 6.8. Rumen liquor (400 ml) was then added to the buffer solution in separate digestion vessels after which CO_2 was purged for 30 s and then sealed. The sealed digestion vessels were placed into the pre-warmed Daisy^{II} incubator. The incubator maintained a constant temperature of 39°C throughout the incubation, while the digestion vessels were continuously agitated. The digestion vessels were removed after 48 h and the filter bags were immediately rinsed for 30 min with cold water to stop microbial activity. After 48 h of incubation, filter bags were removed from the flasks and washed with distilled water. For each treatment, three bags were dried to a constant weight at 105°C, and OMd was obtained by difference following ashing (5 h at 550°C); three bags were used to determine residue N and, finally, three bags were used to determine residue NDF.

Statistical analysis

Data were analyzed by analysis of variance (ANOVA) using CoStat version 1.03 Software (CoHort Software Inc., Monterey, California). The statistical analysis was applied to data following the split-plot design for winter cut forage, and the split–split-plot design for heading stage forage. Differences betweens the means of treatments were compared using the Student–Neumann–Keuls test at $P<0.05$.

Results

DM and protein yield

The ANOVA for combined winter simulated grazing and subsequent cutting at heading did not show significant differences for mycorrhizal fungus inoculation treatment on DMY and protein yield (PY) of triticale forage; therefore, we report the effects of N fertilization and mode of utilization (Table 1). Total DMY (winter cutting + heading cutting), as overall mean of 9.3 t ha^{-1}, increased significantly ($P<0.05$) with N fertilizations (10.7 and 11.4 t ha^{-1}, respectively) compared to no N supplied (5.8 t ha^{-1}). In H modalities, DMY was significantly ($P<0.05$) higher (10.1 t ha^{-1}) than GSH modalities (8.5 t ha^{-1}). Total PY (winter cutting + heading cutting), as overall mean of 0.73 t ha^{-1}, was more than double ($P<0.05$) with fertilization treatment compared with N0, whereas no differences were observed between the modes of utilization of herbage.

Table 4. Effect of mycorrhization and N fertilization on protein fractions of triticale forage at winter cutting.

	Treatment									
	M0			M				Significance		
Item	N0	N100	N100E	N0	N100	N100E	SEM	M	N	M×N
	----------$g\,kg^{-1}$ DM----------									
CP	178	228	191	201	235	221	4.2	ns	*	ns
Protein fractions	----------$g\,kg^{-1}$ CP----------									
A	246	279	261	226	244	231	19.3	*	*	*
B_1	240	262	249	284	307	290	18.7	*	*	*
B_2	281	191	240	255	184	222	13.3	ns	*	ns
B_3	153	198	171	164	205	194	12.1	ns	*	ns
C	81	71	79	71	60	63	2.1	*	*	*

N0, without N fertilization; N100, 100 $kg\,ha^{-1}$ of N, 23 $kg\,ha^{-1}$ of N as diammonium phosphate 18-46-0 before sowing followed by 77 $kg\,ha^{-1}$ of N as ammonium nitrate as top-dressing in February; N100E, 100 $kg\,ha^{-1}$ of N as Entec 25-15-0; M0, without mycorrhiza inoculation; M, with mycorrhiza inoculation.
*$P<0.05$; ns, not significant.
Values are expressed as means of six determinations per treatment.

Forage quality

Data on forage quality of winter and heading cuttings of triticale are reported in Tables 2 and 3, respectively. Regardless of the treatments applied, in the winter cutting the protein content was, as expected, about three times higher than in the forage harvested at heading stage (209 versus 69 $g\,kg^{-1}$ DM, respectively). For the winter cutting, mycorrhizal fungus inoculation significantly lowered ($P<0.05$) ADF content, whereas N fertilization influenced ($P<0.05$) DM, CP, crude fat and ADL contents (Table 2). Conversely, when triticale was cut at heading, the mycorrhizal fungus inoculation treatment resulted in a significant effect ($P<0.05$) on CP, ash, NDF and ADF content (Table 3). Winter cut simulating grazing had a significant ($P<0.05$) effect, increasing NDF and decreasing ADL values of triticale forage harvest in the heading stage. Moreover, the N fertilization improved the forage quality in terms of CP and ADL for both winter and heading cuts. In particular, N100 treatment improved ($P<0.05$) CP content compared to the other N applications (N0 and N100E, respectively) in winter cuts (232 versus 190 and 206 $g\,kg^{-1}$, respectively) and heading cutting (79 versus 60.8 and 67 $g\,kg^{-1}$, respectively). Various interactions among treatments were significant, e.g., CP (G×N) and NDF (M×N, M×G and G×N).

Protein fractionation

The fractionation of CP of winter and heading triticale cuttings are reported in Tables 4 and 5. Mycorrhizal fungus inoculation significantly decreased ($P<0.05$) fractions A (from 262 to 234 $g\,kg^{-1}$ of CP) and C (from 77 to 65 $g\,kg^{-1}$ of CP) and increased fraction B_1 (from 250 to 294 $g\,kg^{-1}$ of CP). The N fertilization had a significant ($P<0.05$) effect, on winter cutting, on all CP fractions. In particular, an increase of fractions A, B_1 and B_3 and a decrease of fractions B_2 and C (Table 4) were observed. This effect was particularly significant ($P<0.05$) in N100 treatments. At the triticale heading stage cut (Table 5), the mycorrhizal fungus inoculation treatment influenced significantly ($P<0.05$) all CP fractions, showing a decrease in fractions A, B_1 and C, and an increase of fractions B_2 and B_3. Winter cut simulating grazing had a significant ($P<0.05$) effect on fractions A and B_2 of the subsequent heading stage cut. N fertilization caused a significant decrease of fractions B_1 and C.

In vitro *digestibility*

Mycorrhizal fungus inoculation treatment did not affect digestibility of winter harvested triticale (Table 6). N fertilization caused a significant improvement in digestibility of CP and NDF in forage ($P<0.05$). There were significant effects of N fertilization and the interaction of inoculation and N fertilization upon digestibility of forage collected at the heading stage (Table 7). However, winter harvested forage was characterized by a decrease in digestibility of CP ($P<0.05$).

Discussion

Triticale, which was not subjected to winter use, led to higher DM production (+19%) compared to the herbage grazed in winter. Decreases in DMY after winter grazing have been related to plant tillering, stem elongation and subsequent growth of leaves and roots[23]. In accordance with our findings, Drake and Orloff[5] reported that delaying initiation of grazing, or re-grazing after 1 to 2

Table 5. Effect of mycorrhization, N fertilization and mode of utilization on protein fractions of triticale forage cut at heading stage.

	Treatment										Significance					
	M0				M											
	H		GSH		H		GSH									
Item	N0	N100	N100E	N0	N100	N100E	N0	N100	N100E	N0	N100	N100E	SEM	M	G	N	M×N	M×G	G×N
	----------g kg^{-1} DM----------																		
CP	60	76	65	60	74	64	63	85	71	60	80	69	1.5	*	ns	*	ns	ns	*
	----------g kg^{-1} CP----------																		
Protein fractions																			
A	227	230	234	250	252	254	200	206	202	221	223	221	18.1	*	*	ns	ns	*	ns
B_1	409	400	403	419	405	410	386	373	370	391	370	381	14.8	*	ns	*	*	ns	ns
B_2	131	130	129	106	118	111	150	154	163	139	152	149	12.7	*	*	ns	ns	*	ns
B_3	153	164	158	145	152	149	185	199	191	190	181	185	10.2	*	ns	ns	ns	ns	ns
C	81	75	77	80	73	76	78	69	74	75	64	70	1.8	*	ns	*	ns	ns	ns

N0, without N fertilization; N100, 100 kg ha^{-1} of N, 23 kg ha^{-1} of N as diammonium phosphate 18–46–0 before sowing followed by 77 kg ha^{-1} of N as ammonium nitrate as top-dressing in February; N100E, 100 kg ha^{-1} of N as Entec 25-15–0; M0, without mycorrhiza inoculation; M, with mycorrhiza inoculation; H, cut at heading stage for hay production; GSH, cut during winter grazing simulation and regrowth cut at heading stage for hay production.

*$P < 0.05$; ns, not significant.

Values are expressed as means of six determinations per treatment.

weeks rest, increases the amount of forage for grazing, but subsequent hay yield and total production are reduced dramatically. However, considering the PY per hectare, GSH is able to provide a protein production similar to H, with the advantage of providing a better distribution of grass during the growing season. Similar results were also obtained by Cazzato et al.[2] in forage mixtures (oat, crimson clover and vetch). The winter grazing resulted in a decrease of the quality of hay produced in terms of NDF content and protein quality, with a decrease of fractions A and B_2; this trend was particularly evident when triticale was not subjected to mycorrhizal fungus inoculation. Presumably, this result was due to the positive effect of mycorrhiza on the plant's ability to overcome stress by cutting in winter, although mycorrhizal fungus colonization of roots was not measured. Lestingi et al.[14] reported that the effect of bio-activators is particularly important when triticale is utilized for grazing and during the first phenological stage. Moreover, despite the decrease of degradable protein, there were no substantial variations in the degree of nutrient digestibility of triticale forage harvested at the heading stage in response to simulated grazing. The mycorrhizal fungus inoculation had a positive effect on the quality of the forage harvested at heading stage which resulted in less fiber and more protein. The presence of more protein in inoculated plants was mainly due to their higher uptake of N from the soil[24]. The inoculated plants had also less NDF and ADF content. The data pertaining to the decrease in protein content and increase in NDF and ADF with increased in DM of plants confirm the findings of Reling et al.[25]. It is also reported by Sukhchain and Sidhu[26] that fodder quality in general tends to decline as DMY improves. Although N fertilizer improved CP in plants, it also enhanced NDF and ADF. This is also corroborated by Bamikole et al.[27]. In our study, DMY and CP content increased in inoculated plants, but NDF content remained low. This may be interpreted as biofertilizers being a more suitable source of fertilizer in grasses than chemical fertilizer to improve their overall production and fodder quality. Our findings are in agreement with those of Martínez-López et al.[28], who found significant improvement of forage when inoculated with mycorrhiza in terms of higher CP and lower NDF contents. Few investigations have been conducted on the effect of mycorrhiza application on the protein fractions in cereal and forage crops. In this field trial, when mycorrhiza were applied, the forage protein quality of winter grazing improved because it contained less indigestible protein (fraction C) and NPN (fraction A), and a positive increase in medium-degradable fractions in cutting hay. These results indicate that the use of plant inoculation could modify the distribution of protein fractions in triticale forage. In contrast to our findings, Lestingi et al.[14] found that the utilization of bio-activators increased, even if not significantly, fraction A compared with fraction B_1, due to different nitrate organization capability in triticale

Table 6. Effect of mycorrhization and N fertilization on *in vitro* digestibility of triticale forage at winter cutting.

	Treatment									
	M0			M				Significance		
Item	N0	N100	N100E	N0	N100	N100E	SEM	M	N	M×N
	----------$g\,kg^{-1}$ DM----------									
DMd	487	492	489	501	509	503	3.7	ns	ns	ns
	----------$g\,kg^{-1}$ OM----------									
OMd	530	527	530	550	542	549	4.1	ns	ns	ns
	----------$g\,kg^{-1}$ CP----------									
CPd	491	517	512	503	521	511	4.8	ns	*	*
	----------$g\,kg^{-1}$ NDF----------									
NDFd	388	412	401	399	422	408	4.2	ns	*	ns

N0, without N fertilization; N100, 100 $kg\,ha^{-1}$ of N, 23 $kg\,ha^{-1}$ of N as diammonium phosphate 18-46-0 before sowing followed by 77 $kg\,ha^{-1}$ of N as ammonium nitrate as top-dressing in February; N100E, 100 $kg\,ha^{-1}$ of N as Entec 25-15-0; M0, without mycorrhiza inoculation; M, with mycorrhiza inoculation.
*$P<0.05$; ns, not significant.
Values are expressed as means of 12 determinations per treatment.

at different phenological stages. The lack of consistency with results of the previous study may be partially attributed to the arbitrary dates for forage removal, which in many cases did not take into account the stage of development of the crop, the different pedological condition of the study site and the agronomic techniques.

N fertilization, as the average of the two N application modalities, caused an increase in DMY (+90%) and in protein production (+130%), in agreement with results found by Cazzato et al.[2] and Harmoney and Thompson[29]. Our result could be attributed to the reduced content of N in the soil. Fertilization has a direct effect on the proportion of plant N present as true protein[30]. N fertilization consistently caused an increase of CP, a decrease of lignin and, in terms of protein quality, an increase in NPN only in winter forage; this trend was particularly significant when the N was split into two phases (N100), presumably because the winter cutting was performed about 3 weeks after top dressing. In fact, the N uptake by the plant increased with the level of N fertilization and this led to a build-up of non-protein organic N, thereby decreasing the proportion of protein N. At high levels of fertilization, the greater N uptake was followed by an accumulation of nitrate in the plant which further reduced the protein N fraction. The nitrate content of grass reached a maximum 2 weeks after N application, as found by Wilman[31]. The reduction of absorbed nitrates into ammoniacal N and its incorporation into protein can, in fact, be considered as a single step limited by the enzyme nitrate reductase that mediates the first reaction[30]. The application of N fertilization tended to decrease the fraction of indigestible protein (fraction C) both in fresh forage and in forage cut at heading stage. This result could explain the moderate improvement of protein and fiber digestibility of forage produced in our trial. According to the effect of N fertilization on the CP content of grass, cell wall digestibility may have been lowered on less fertilized swards. Peyraud et al.[32] reported a decrease of fiber digestibility by 0.06 unit when the CP content of grass decreased from 160 to 110 $g\,kg^{-1}$ DM, whereas no variation in fiber digestibility was observed when the CP content of grass decreased from 210 to 170 $g\,kg^{-1}$ DM[33]. This may have been related to conditions prevailing within the rumen when cellulolytic activity was impaired by a shortage of degradable N supply[34,35], or it may have been related to inherent characteristics of the plant cell wall. Messman et al.[36] and Valk et al.[37] indicated that there is a lower rate of degradation of cell walls with reduced levels of N fertilization. This effect on cell wall digestibility may explain why greater variations in OM digestibility have been reported when comparing with extreme reductions in N fertilization. For example, Reid et al.[38] reported a decrease of 0.05 unit in OM digestibility for a reduction of 500–0 kg $N\,yr^{-1}$. Finally, the moderate effect of reduced N fertilization on OM digestibility of grasses could be attributed to the fact that any decrease in the CP content of forage is partially compensated for by an increase in water soluble carbohydrates which are highly digestible, despite the risk of a moderate decrease in fiber digestibility.

Conclusions

When triticale was grown for forage in a Mediterranean climate, mode of utilization and N fertilization affected forage yield and quality, while mycorrhizal fungus inoculation influenced positively some quality parameters. The utilization of herbage cut at heading stage improved DMY, therefore this practice is recommended when the farm management does not provide grazing

Table 7. Effect of mycorrhization, N fertilization and mode of utilization on *in vitro* digestibility of triticale forage cut at heading stage.

	Treatment										Significance					
	M0				M											
	H		GSH		H		GSH									
Item	N0	N100	N100E	N0	N100	N100E	N0	N100	N100E	SEM	M	G	N	M×N	M×G	G×N
	---g kg^{-1} DM---															
DMd	501	524	511	498	515	508	512	525	520	519	530	537	3.9	ns	ns	*	*	ns	ns
	---g kg^{-1} OM---															
OMd	530	557	550	522	549	531	559	575	560	558	579	570	4.6	ns	ns	*	*	ns	ns
	---g kg^{-1} CP---															
CPd	528	559	540	521	541	532	547	562	551	535	560	541	3.7	ns	*	*	*	*	ns
	---g kg^{-1} NDF---															
NDFd	467	488	444	431	459	444	434	460	448	465	490	472	4.8	ns	ns	*	ns	ns	ns

N0, without N fertilization; N100, 100 kg ha^{-1} of N, 23 kg ha^{-1} of N as diammonium phosphate 18-46-0 before sowing followed by 77 kg ha^{-1} of N as ammonium nitrate as top-dressing in February; N100E, 100 kg ha^{-1} of N as Entec 25-15-0; M0, without mycorrhiza inoculation; M, with mycorrhiza inoculation; H, cut at heading stage for hay production; GSH, cut during winter grazing simulation and regrowth cut at heading stage for hay production.
*$P < 0.05$; ns, not significant.
Values are expressed as means of 12 determinations per treatment.

animals, whereas winter grazing offers the advantage of a better grass distribution without reducing PY. N fertilization was very effective regardless of mode of herbage utilization due to its positive effects on forage production and quality in terms of CP and nutrient digestibility. In addition, splitting N fertilization enhanced the protein quality only in fresh forage compared to a N fertilizer with NI applied just once before sowing. Finally, the results about the improvement of forage quality with mycorrhizal fungus inoculation suggest the need to study this issue in depth.

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Urine as an alternative fertilizer in agriculture: Effects in amaranths (*Amaranthus caudatus*) production

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Abstract

Crop production in most developing countries is faced with a dearth of resources for optimum production of which fertilizer is one. The use of human urine as well as its mixture with compost are potential solutions to this problem. Thus, this report investigated the influence of human urine and its combination with compost on yield and soil quality of land under green amaranths (*Amaranthus caudatus*). This study involved a field experiment to determine the response of green amaranths to the application of 100% urine, 2/3 urine N + 1/3 compost N, 100% compost N, NPK (15:15:15) at the rate of 100 kg N ha^{-1} and control with no fertilizer treatment using farmers' participatory approach. The vegetables produced from the experimental treatments were analyzed in the laboratory for pathogenic microbial risk as well as effects of the fertilizer on nutrient status of the experimental soils (before and after planting). Perception of farmers and consumers in the study area regarding use of urine as fertilizer for vegetable production was investigated with the aid of a structured questionnaire. The result of this investigation revealed that 100% urine resulted in 58.17 t ha^{-1} total plant yield, while NPK 15:15:15 gave 34.34 t ha^{-1} total plant yield in the two plantings. Microbial analysis of edible portion of vegetable from plot fertilized with urine did not reveal any significantly different pathological contamination compared to other fertilizer treatments used in this investigation. Urine treatment improved soil nutrient exchangeable cations and acidity. The perception study revealed that respondents perceived urine to be a good agricultural input that could be used as a fertilizer in vegetable crop production and there was no strong cultural norm that would prevent them using it for crop production. Vegetable consumers would also buy vegetable crops grown with urine if they are well informed about its safety for crop production. Since the use of urine as fertilizer for crop production improved amaranth's yield and did not show any negative implication on soil environment, human urine seems to have good potential both in crop yield and acceptability by farmers and consumers.

Key words: human urine, amaranths, fertilizer, perception, nutrient status, crop yield

Introduction

Rapid urbanization will result in the generation of more wastes in cities including human urine which has been found to be a good fertilizer for crop production[1,2]. However, economic returns and microbial contamination as well as acceptability are issues of concern.

Physiological measurement shows that an adult in a year excretes via urine 2.5–4.3 kg N, 0.7–1.0 kg P and 0.9–1.0 kg K. This is higher than levels excreted via feces and represents 60–90% of the plant N, P and K ingested, which can be retrieved in solution[3]. Although excellent performance of human urine as a fertilizer in crop yield has been reported[4–6], there is a need to quantify financial returns from urine fertilizer.

It has been reported that fresh human urine is sterile in the bladder[7]. *Leptospira interrogans*, *Salmonella typhi*, *Salmonella paratyphi* and *Schistosoma haematobium* are pathogens traditionally known to be excreted in urine[8]. The possibility of transmitting these pathogens via urine as fertilizer in developed countries is negligible[7]. Where human wastes such as urine can improve agricultural productivity, independently of their microbial characteristics, they can contribute to the nutritional status of the population, thus improving public health[4]. However, it is necessary to investigate whether the same will hold in a

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Figure 1. Satellite imagery of the project area, Mokola, Ibadan, Nigeria. The arrow points to the experimental plot.

developing country such as Nigeria with less strict environmental health policies. Although the use of urine in agriculture is not new[9], attitudes of farmers and consumers to acceptability in crop production is not certain.

Richert *et al.*[10] reported that dissemination and knowledge development on urine as a fertilizer are best gained through local demonstration experiments involving organizations that work with small-scale farmers and local communities, as well as local research organizations. Thus, this report presents an investigation of the effects of human urine in comparison with other fertilizer materials on the yield of amaranths (*Amaranthus caudatus*) on small-scale farms. Post-production analysis of soil nutrient and crop pathogen concentration was done as well as the study of the perception of farmers and consumers to the use of urine in crop production.

Materials and Methods

This study involved a field experiment to determine the response of green amaranths (*A. caudatus*) to the application of urine and other fertilizer sources using a participatory approach. The vegetables produced from the experimental treatments were analyzed in the laboratory for pathogenic microbial risk as well as the effects of the fertilizer on the nutrient status of the experimental soils before and after planting. Perceptions of farmers and consumers in the study area regarding use of urine as fertilizer for vegetable production was investigated with the aid of a structured questionnaire, participatory field planting/demonstration and laboratory analyses.

This investigation was carried out between December 2008 and August 2009. The study was carried out at the Mokola-Army Barracks of Ibadan, North West Local Government, Oyo State, Nigeria. The local government owns a large area of land with a population of about 147,918 according to the 1991 Census. Average annual rainfall is 1280 mm with a bimodal pattern. Minimum and maximum temperatures range from 12 to 30°C and 28 to 34°C, respectively.

Commercial vegetable production in urban centers in Nigeria is dependent on cultivation of institutional land (such as the Army Barracks). There are up to five such vegetable farming communities within urban and peri-urban Ibadan, but the project community was selected due to the high level of cooperation from the farming community. Farmers in this area are mostly men from different ethnic groups of Nigeria. An aerial photograph of the study location is shown in Figure 1.

Agronomic study

The planting experiment involved cultivation of green amaranths with fertilizer treatments at a rate of 100 kg N ha^{-1} using 100% urine, 2/3 urine N + 1/3 compost N, 100% compost N, NPK (15:15:15) and control with no fertilizer treatment. Urine was collected from a male hostel at the University of Ibadan and some households in Ibadan, while the compost is a commercial product. The collected urine was stored under airtight conditions for a month. The total land area is 63 cm^2 (average size of small-holding farm in the study area) made up of 20 beds of 2 m × 1 m with fertilizer treatments replicated four times. Plant population was at a rate of 1.8 million plants ha^{-1}. The experiment was laid in a randomized complete block design. Planting was done in two successions; first planting with fertilizer application and second planting without fertilizer application (residual). Yield parameters observed were subjected to statistical

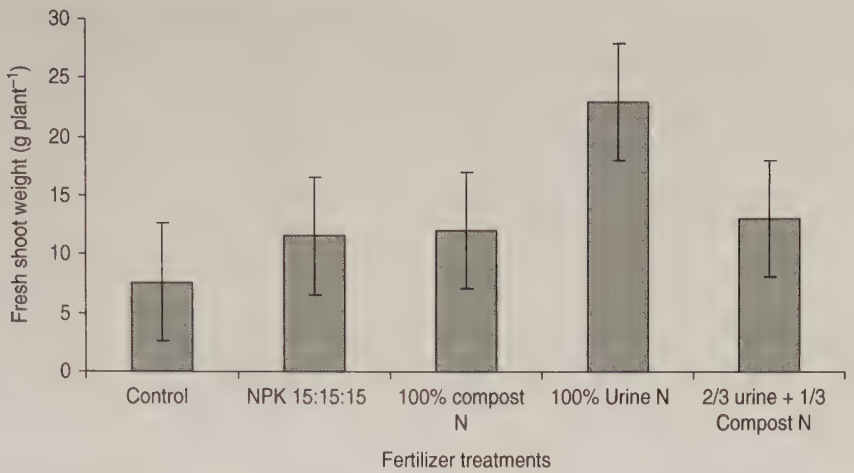

Figure 2. Effects of fertilizer treatments on fresh shoot weight yield of green amaranths at 4 weeks after the first planting.

analysis using least significant difference to separate means.

Soil samples were collected at depths of 0–15 cm using a soil auger before the experiment began. The physical and chemical analysis of the experimental soils was carried out before the treatments were added to the soil. The soil was air-dried and sieved using a 2 mm sieve. Soil particle analysis was determined by mechanical analysis, using the hydrometer method[11]. Soil pH was determined in water and KCl (both in ratio 1:1) using a pH meter with a glass electrode. Total nitrogen was determined using the macro-Kjeldahl procedure[12], while organic carbon was determined by using the wet oxidation method[13]. Organic matter content was determined by multiplying the organic carbon content by 1.74, while available phosphorus was determined by using the Bray1 method[14]. Exchangeable potassium (K), calcium (Ca), magnesium (Mg) and sodium (Na) were extracted with 1 N ammonium acetate. The concentration of K, Ca and Na in the filtrate was determined using a flame photometer, while Mg was determined using a Perkins-Elmer atomic absorption spectrophotometer. Exchangeable acidity was determined by means of the titration method and effective cation exchange capacity was determined by summation of exchangeable bases. Extractible copper, zinc, iron and manganese were extracted with EDTA and determined with an atomic absorption spectrophotometer.

Microbial analysis

The microbial analysis was performed on both main and residual plantings of green amaranths from the fertilizer trials on the field. Vegetable samples were selected randomly and taken to the laboratory in sealed envelopes to prevent contamination. An edible portion of the crop from each treatment was homogenized and the fluid extracted and serially diluted to obtain a 10^{-3} dilution factor using the ten-fold dilution method using sterilized nutrient agar for the isolation of aerobes and sterilized MacConkey agar was used for isolation of the coliforms/enterobacteria.

Perception of farmers and consumers on use of urine as fertilizer

Data were collected using the interviewer-administered questionnaires to obtain information from both the farmers working on the vegetable farm and the household members living in the barracks on socio-demographic characteristics, hygiene and sanitation practices and method of waste disposal. A total of 161 respondents were randomly interviewed for the perception study, of which 60 were farmers and the remaining 101 were consumers or retailers. A structured questionnaire was used to collect data. The interview schedule was conducted on a face-to-face basis with the respondents, while those who could fill in the questionnaires were given the opportunity to do so, with little or no assistance.

Results

The objectives of this project were to investigate the influence of human urine and its compost mixture on the yield of amaranths (*A. caudatus*), assess their impact on soil and crop quality as well as attitudes of farmers and consumers toward the use of urine in crop production.

Response of amaranths to urine, municipal city waste compost mixture

The 100% urine N-treated plants had the highest significant ($P<0.05$) fresh total plant weight of 23 g plant^{-1} (41.4 t ha^{-1}) at the first planting (Figure 2). The lowest yield of 7.6 g plant^{-1} was recorded from the control plot (no treatment applied) and this yield was not significantly different from other treatments, except that of 100% urine. The results of yield parameters revealed better performance of organic sources on the plants as compared

Table 1. Estimated partial income from *A. caudatus* production for two successive plantings.

Treatments	Fresh total yield g plant^{-1} at planting periods			Yield (t ha^{-1})	[2]Gross income (NGN ha^{-1})
	1st	2nd[1]	Total		
Control	7.6b	5.3	12.9	23.22	928, 800
NPK	11.2b	7.88	19.08	34.34	1,373,600
100% compost	12.1b	5.97	18.07	32.53	1,301,200
100% urine	23a	9.32	32.32	58.17	2,326,800
2/3 urine + 1/3 compost	13.6ab	5.25	18.85	33.93	1,357,200

Means with same letters within column are not significantly different by Duncan's Multiple Range Test ($P = 0.05$).
[1] No significantly difference in mean by Duncan's Multiple Range Test ($P = 0.05$) within the column.
Yield at 40 (farm-gate value) kg^{-1}.
[2] Partial income (without other costs of production).
1 = 150 NGN as on 25 January, 2011.

to mineral fertilizer NPK. The best performance was seen with 100% urine N. A similar trend was observed during the second planting season (residual) for 100% urine N (see Table 1). Estimated partial gross return from the production of amaranth revealed that 100% treatment produced the highest value of 2,326,800 NGN, while the least (928,800 NGN) came from control (Table 1). The effect of the applied treatments on plant stem girth is reported in Table 2. The 100% urine N treatment also resulted in the best stem girth (2.57 cm), followed 100% compost N, while there was no significant difference between the control and NPK treatments.

Micro-organisms isolated and identified in vegetable samples

Coliforms isolated from edible portions reveal a slight difference in the presence of *Escherichia coli* in samples with 100% urine, 2/3 urine and 1/3 compost and mineral NPK-treated samples from the first planting (Table 3). However, the second (residual) harvest did not show presence of *E. coli*. Also, the number of organisms isolated from the residual planting reduced as compared to the first planting. As shown in the table, the main planting had three basic aerobes isolated from the vegetables. While *Bacillus* spp. and *Staphylococcus* spp. was common to all the samples but samples from 2/3 urine + 1/3 compost, 100% compost and Mineral NPK-treated plots had *Pseudomonas* spp., which was absent in the 100% urine and control treatments. The same aerobes were isolated from main and residual plantiings.

Table 4 shows the total coliform and aerobic microbial species count from the harvested amaranths in the first and second planting. The result reveals that samples from control, 100% urine and 2/3 urine + 1/3 compost had highest coliform counts as compared with NPK and 100% compost treatments in the first planting harvest. The lowest coliform count was recorded in vegetables fertilized with 100% compost. At the second planting harvest, urine treatment samples had the highest coliform count. Generally, the coliform count was much reduced in the second planting as compared with the first. Total aerobic count from the first planting harvest followed the same trend in the coliform count, with control and urine samples with counts of 2.9×10^5 having the highest population of aerobes, followed closely by 2/3 urine + 1/3 compost sample. At the second plant harvest, all samples had the same value of 3.0×10^4 aerobic count.

Table 2. Effect of fertilizer treatments on stem girth (cm) of *A. caudatus*.

Treatments	Stem girth (cm)	
	1st planting	2nd planting
Control	1.85	1.725
NPK	2.02	1.875
100% compost	2.27	1.775
100% urine	2.57	1.950
2/3 urine + 1/3 compost	2.10	1.650
S.E.D. ($P < 0.05$)	0.42	Ns

S.E.D. = Standard errors of differences of means ($P < 0.05$).
Ns = means difference not significant.

Effects of the applied treatments on soil nutrient properties

Generally, the soil used is acidic with a pH of 4.8. At the end of the experiment 100% urine and urine plus compost plots had their pH reduced to 4.7. The increase in soil organic matter was also noticed in the treated plots. The 100% urine plots had the highest organic carbon 13.08 g kg^{-1} followed by the other two organically treated plots. Effects of the applied treatments on other soil essential nutrient elements determined were not that different (Table 5).

Perception of farmers on the use of urine as fertilizer for crops

Social characteristics of farmers. In general, the majority of the respondents (farmers) in this area of investigation were Christians (i.e., two-thirds). This shows that more Christians are involved in vegetable

Table 3. Micro-organisms isolated and identified in harvested amaranths.

	100% urine	Control	2/3 Urine and 1/3 Compost	100% compost	Mineral NPK
First planting					
Coliforms					
	Aeromonas hydrophilia	*A. hydrophilia*	*Escherichia coli*	*A. hydrophilia*	*Klebsiella pneumonia*
	Enterobacter aerogenes	*E. aerogenes*	*E. aerogenes*	*E. aerogenes*	*E. coli*
	E. coli	*Proteus vulgaris*	*P. vulgaris*	*Pseudomonas aeroginosa*	*Pseudomonas cepacia*
	K. pneumonia	*Proteus mirabilis*	*P. mirabilis*	*P. mirabilis*	*P. mirabilis*
	P. aeroginosa	*Pseudomonas putida*	*P. putida*	*P. putida*	
Aerobes					
			Pseudomonas spp.	*Pseudomonas* spp.	*Pseudomonas* spp.
	Bacillus spp.	*Bacillus* spp.	*Bacillus* spp.	*Bacillus* spp.	*Bacillus* spp.
	Staphylococcus spp.	*Staphylococcus* spp.	*Staphylococcus* spp.	*Staphylococcus* spp.	*Staphylococcus* spp.
Second planting					
Coliforms					
	A. hydrophilia			*E. aerogenes*	
	P. mirabilis	*P. vulgaris*	*P. mirabilis*	*P. vulgaris*	*P. mirabilis*
	P. cepacia	*A. hydrophilia*	*P. putida*	*P. cepacia*	*P. cepacia*
Aerobes					
	Bacillus spp.	*Bacillus* spp.	*Bacillus* spp.	*Bacillus* spp.	*Bacillus* spp.
	Pseudomonas spp.	*Pseudomonas* spp.	*Pseudomonas* spp.	*Pseudomonas* spp.	*Pseudomonas* spp.
	Staphylococcus spp.	*Staphylococcus* spp.	*Staphylococcus* spp.	*Staphylococcus* spp.	*Staphylococcus* spp.

production than Muslims in the study areas. Eighty-five percent of the respondents reported they have no problem getting land, as long as they could make the proper request for land from the concerned authority.

Farmers' perception and willingness to use urine as a fertilizer. According to Table 6, most of the respondents (69%) had not heard that urine is also a fertilizing agent that can be used for vegetable crop production. Most of the respondents (47%) perceived urine as an organic material, though they never knew it could be incorporated into vegetable crop cultivation. Thirty-five percent perceived it as a good agricultural innovation. A few (7%) felt that it is taboo and not hygienic (28%) to use urine for vegetable crop production. Among the respondents, 77% stated that they do not have any cultural norms against the use of urine for vegetable crop planting. Also, 43% of the respondents felt that urine is an organic matter that could be a good agricultural innovation, while 8% believed that it could be easily available, while only 5% have the opinion that urine would be cheaper. More than two-thirds of the respondents did not have any opinion on religious beliefs against the use of urine for vegetable crop production. Also, 63% (no response) of the respondents did not specify any hindrance in economic terms to the use of urine.

Consumers' perception and willingness to buy vegetables planted with urine as fertilizer

The perception of consumers on willingness to buy vegetables produced with urine fertilizer is reported in Table 7.

Table 4. Counts of total coliform and aerobic species (cfu ml^{-1}) isolated from harvested amaranths.

	Total coliform spp. (cfu ml^{-1})		Total aerobic spp. (cfu ml^{-1})	
Treatments	**1st planting**	**2nd planting**	**1st planting**	**2nd planting**
Control	2.5×10^5	2.5×10^4	2.9×10^5	3.0×10^4
NPK	7.0×10^4	2.7×10^4	1.7×10^5	3.0×10^4
100% compost	2.1×10^4	2.3×10^4	1.2×10^5	3.0×10^4
100% urine	2.5×10^5	3.0×10^4	2.9×10^5	3.0×10^4
2/3 urine + 1/3 compost	2.1×10^5	2.0×10^4	2.6×10^5	3.0×10^4
Mean	19.5×10^4	2.5×10^4	22.6×10^4	3.0×10^4
Min	7.0×10^4	2.0×10^4	12.0×10^4	3.0×10^4
Max	2.5×10^5	3.0×10^4	29.0×10^4	3.0×10^4
SD	8.5×10^4	3.8×10^3	7.7×10^4	0

Sixty-two percent of the vegetable consumers were indifferent to the use of urine as fertilizer in producing vegetables, while just 8% considered urine use in crop production as forbidden by their religions. A greater proportion (24%) of the consumers would buy vegetables produced with urine fertilizer if urine does not pose any health problem, while 29% would purchase the vegetables if they do not smell urine. Only 9% of the consumers indicated interest in buying the vegetables even if nutritional value is affected by the urine fertilizer considering that urine is readily available as fertilizer for crop production.

Table 5. Chemical properties of soil at the end of second planting of amaranths.

Treatments	pH	OC ($g\,kg^{-1}$)	N	P ($mg\,kg^{-1}$)	K	Ca	Mg	Na	Ex. acidity	ECEC ($cmol\,kg^{-1}$)	Fe	Mn	Cu	Zn ($mg\,kg^{-1}$)
Pre-planting soil	4.8	8.64	0.9	14	0.4	5.2	1.0	1.0	0.6	8.2	109	179	22	66
1st planting														
Control	4.9	10.44	1.3	13	0.2	6.5	0.4	0.9	0.4	8.4	104	127	33	104
NPK	4.9	12.96	1.7	16	0.2	6.5	0.4	0.9	0.2	8.1	118	142	36	103
100% compost	4.8	11.16	1.8	13	0.2	6.8	0.3	0.9	0.6	8.7	146	102	36	86
100% urine	4.8	12.48	2.4	14	0.2	7.1	0.4	1.0	0.4	9.0	103	126	37	101
2/3 urine + 1/3 compost	4.9	13.68	2.9	14	0.2	7.1	0.4	1.0	0.4	9.1	104	115	35	97
2nd planting														
Control	4.8	11.64	2.5	15	0.3	7.5	0.7	1.2	0.4	10.1	108	170	32	110
NPK	4.8	11.28	1.6	17	0.3	7.6	0.7	1.2	0.4	10.2	104	147	30	90
100% compost	4.8	12.96	2.0	15	0.3	7.9	0.7	1.2	0.4	10.5	106	143	34	96
100% urine	4.7	13.08	1.8	15	0.3	7.6	0.6	1.2	0.2	9.9	108	168	33	104
2/3 urine + 1/3 compost	4.7	12.36	1.9	15	0.3	7.5	0.6	1.3	0.2	9.8	103	160	34	104
Means	4.8	11.88	1.9	15	0.3	7.0	0.6	1.1	0.4	9.3	110	144	33	96
Min	4.7	8.64	0.9	13	0.2	5.2	0.3	0.9	0.2	8.1	103	102	22	66
Max	4.9	13.68	2.9	17	0.4	7.9	1.0	1.3	0.6	10.5	146	179	37	110
Std	0.1	1.45	0.6	1	0.1	0.8	0.2	0.1	0.1	0.9	13	24	4	12

Std = standard deviation.

Discussion

Amaranthus caudatus is usually consumed fresh, thus the emphasis on fresh weight of the plants. The agronomic experiment revealed that the application of 100% urine N resulted in the highest significant ($P<0.05$) fresh total plant weight of $23\,g\,plant^{-1}$ ($41.4\,t\,ha^{-1}$) at the first planting. This yield is far higher than $20\,t\,ha^{-1}$ on a sandy soil of poor fertility status[15]. A similar yield of $16.78\,t\,ha^{-1}$ on similar soils was also reported[16]. Higher and better yield in 100% urine N-treated plants must have been due to the fact that nutrients in urine are in forms that are readily available to plants. The urea in urine readily degrades to ammonium and nitrate ions, both of which are in forms that plants can absorb[7]. The improvement on amaranth yield by the 100% urine N over that of mineral fertilizer NPK is in agreement with reports of other researchers in some parts northern Europe[17,18].

A combination of yields ($t\,ha^{-1}$) from both planting seasons resulted in 100% urine-treated soil producing $58.17\,t\,ha^{-1}$ total plant mass, while NPK 15:15:15 gave $34.34\,t\,ha^{-1}$ total plant yield. It was observed that the estimated partial gross returns of 2,326,800 NGN attributable to 100% urine fertilizer doubled that from the use of mineral fertilizer NPK. This is an indication that urine fertilizer could be profitable.

The results obtained agree with research findings from Uganda where urine was found to increase yield more than mineral fertilizer NPK[5]. It also fits with results from Finland where cabbages grown with human urine performs better than those from conventional plots[6]. Similar results have been seen from previous experiments carried out in the University of Ibadan on *Celosia argentea*[19].

Generally, urine- and compost-treated soils were better in nutrient status at the end of the second planting. This increase in organic carbon might be associated with the increase in microbial activities usually found in fields where organic fertilizers are used[20]. Organic carbon content of the NPK plots was reduced at the end of the experiment (organic carbon at the end of the first planting was $12.96\,g\,kg^{-1}$ and reduced to $11.28\,g\,kg^{-1}$ at the end of the second planting). Total nitrogen of the soil increased from 1.3 up to $2.0\,g\,kg^{-1}$ on treated plots. The least improvement as found on NPK plots ($1.6\,g\,kg^{-1}$). This indicates that urine and compost fertilizers can improve soil nutrient status.

The same populations of aerobes were isolated from main and residual plantings. Generally, high levels of bacteria were recorded in vegetables which were higher than the recommended level of $<10^3\,cfu/100\,g$[21]. This situation could have been caused by active organisms present in the soil/plant layer interface rather than urine or compost applied. Several factors may account for the high levels of coliform contamination recorded in most of the analyzed vegetables. The principal possible source of the coliform is the application of uncured poultry droppings as fertilizers by the farmers in the investigated area. Fresh poultry litter samples, sometimes used without sufficient drying for vegetable production in Kumasi, had equally high fecal coliform counts between 3.6×10^4 and $1.1\times10^7\,cfu/100\,g$.[22]

Comparatively lower levels of bacteria recorded in compost-grown vegetables were indicative of the

Table 6. Farmers' perception and willingness to use urine as fertilizer.

Parameters	Frequency	Percentage
Knowledge of urine use in agriculture		
Yes	17	28
No	38	63
Indifferent	5	8
Total	60	100
General believe about urine in agriculture		
Possible source of organic fertilizer	28	47
Cheapness	6	10
Ready availability	6	10
Good innovation	20	33
Total	60	100
Personal opinion against use of urine as fertilizer		
Bad smell	23	38
Irritating to handle	13	22
A taboo	4	7
Not hygienic	17	28
Indifferent	3	5
Total	60	100
Existence of cultural norms against use of urine		
Have	14	23.30
Do not have	46	76.70
Total	60	100
Respondents' personal conviction on urine use in agriculture		
Organic fertilizer	26	43
Good innovation	26	43
Would be cheaper	3	5
Availability	5	8
Total	60	100
Specific religious beliefs against urine usage		
Personal reason	6	10
It is forbidden	3	5
Its purity	8	13
No response	43	72
Total	60	100

Source: Field survey, 2009.

Table 7. Consumers' perception and willingness to buy vegetables planted with urine as a fertilizer.

Parameters	Frequency	Percentage
Personal and religious belief against urine usage		
Personal reason	14	14
Its forbidden	8	8
It is impurity	16	16
Indifferent	63	62
Total	101	100
Conditions for buying vegetables grown with urine		
If substitutes are scarce	26	26
If substitutes are available but the vegetables from urine fertilizer are cheaper	18	18
If substitutes are scarce but vegetables are succulent and leafy	14	14
If substitutes are available but the vegetables are fresh and succulent	20	20
If the vegetables are not smelling of urine	29	29
If the vegetables are smelly and odor could be washed away by boiling	23	23
If urine does not pose any health problem	36	36
If it poses health problem but cheaper	18	18
If the nutritional value is not affected	24	24
If nutritional value is affected but readily available	9	9
If no extra cost is incurred in use of urine	11	11
If extra cost is added but vegetable is fresh and succulent	10	10
Total	101	100

Source: Field survey, 2009.

fact that the compost was highly cured and had less bacterial contamination. It was possible that microbes in the urine samples also contaminated the urine/compost composite mixture used in growing vegetables hence the high levels of microbes recorded. In general, the results of the microbial analysis did not establish any negative pathological contamination due to the use of urine. This result is in line with earlier reports that the use of properly treated urine as a fertilizer does not introduce pathological contamination to agricultural fields[7,21].

The perception analysis revealed that most farmers and consumers do not have serious cultural norms against urine fertilizer. Analysis showed that most farmers would use urine for vegetable production if it gives a better yield than other organic fertilizers, while most consumers would buy vegetable crops grown with urine if it does not pose any health problem. These results are similar to a report that people in Eastern Uganda's Mbale District accepted the use of urine as a fertilizer because of the financial gain attached[23].

Conclusion

The results of the response of *A. caudatus* to urine and municipal waste composts and their mixtures as fertilizer revealed that 100% urine performed better as a fertilizer at both main and residual plantings. Financial returns using urine as a fertilizer also doubled that of the commonly used mineral fertilizer NPK. The use of urine as fertilizer for *A. caudatus* improved some useful soil properties and did not result in pathogenic microbial contamination. Although coliform and aerobic bacterial species were detected in the vegetable, this could not be traced to the application of urine or other treatments used in this study but to the poor soil management practices by the farmers before the investigation. Respondents in this study perceived urine to be a good agricultural innovation which can be used for vegetable crop production and did not have any strong cultural norms that could prevent

them from using urine for planting or that would prevent them from buying vegetable crops grown with urine. This is an indication that if urine is harvested and made available, farmers would be willing to use it for crop production as long as consumers are willing to buy. However, the logistics of storage and distribution of urine as fertilizer have to be addressed.

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Insecticide use and crop selection in regions with high GM adoption rates

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Abstract

South Dakota has been a leading adopter of genetically modified organism (GM) crops since their introduction in 1996. In 2009, South Dakota shared the top adoption rate with Iowa for the percentage of acres planted with Bt corn. However; South Dakota has also recently experienced a significant increase in the proportion of acres treated with insecticide. The empirical evidence presented suggests that corn, hay and sunflower production in South Dakota have experienced an intensification of insecticide use in 2007 relative to past US Census of Agriculture reporting years. This study links the proportion of acres planted for a specific crop to the proportion of total acres treated with insecticide at the county level. This approach provides insight on how changing cropping patterns in South Dakota have influenced insecticide use. Empirical results indicate that the upper-bound estimate for insecticide usage on non-Bt corn acreage increased from 38% in 2002 to all non-Bt corn acres planted in 2007. The implication of this result is that in 2007 South Dakota producers were likely treating a percentage of their Bt corn acres with insecticide. Changing cropping patterns in South Dakota are also compared to that in other states in the US Corn Belt region. It appears that the South Dakota experience is not unique and is part of a broader trend.

Key words: insecticides, genetically modified organism, targeted insect pests, crop production, pest management practices

Introduction

South Dakota lies in the northern corn belt of the United States and agricultural production is a main driver of South Dakota's economy. In 2008, agriculture accounted for close to 10% of South Dakota's $37 billion economy[1]. The agricultural industry in the state has undergone a myriad of changes that are related to the rapid adoption of biotechnology and an increased demand for corn driven by the rapid expansion in corn-based ethanol production in South Dakota[2]. These two factors have contributed to a change in the crop rotation profile over the last 30 years, shifting from a profile with a large proportion of wheat, barley and oats toward one that is dominated by corn and soybeans. In 2010, South Dakota was ranked: (a) seventh in corn production, (b) eighth in soybean production, (c) second in sunflower production and (d) fourth in hay production in the United States[3].

South Dakota has adopted genetically modified (GM) varieties faster than any other state (see Fig. 1)[4]. The rapid adoption of GM crops in South Dakota over the past 15 years suggests that a change in pest management practices should have occurred during this period. One of the predicted benefits of the adoption of GM crops is a decline in insecticide use due to the ability of insecticidal GM crops to kill specific target insects.

In South Dakota, corn acres planted with Bt or stacked gene varieties increased from 37% in 2000 to 71% in 2009[5]. In 2009, South Dakota shared the top adoption rate with Iowa for Bt and stacked gene varieties. Even though South Dakota is the highest adopter of this type of technology, there has been an increase in the number of acres and the proportion of acres planted that have been treated with

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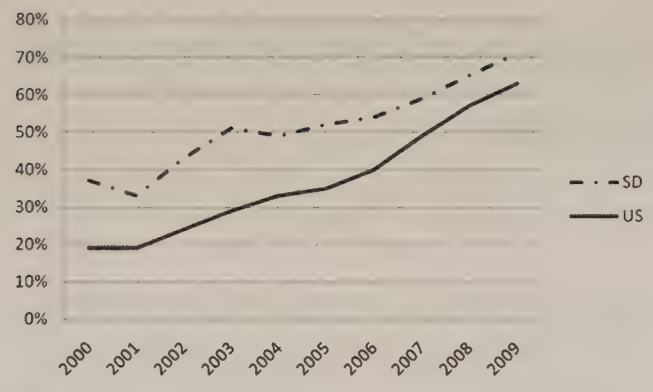

Figure 1. Adoption of insect-resistant and stacked GM varieties for South Dakota and the US (data collected from USDA-ERS website[4]).

insecticide over the past two decades. This positive trend has been gradual from 1978 to 2002, but in 2007 the increase was significantly above the trend. In 2002, South Dakota producers treated a total of 1.3 million acres with insecticides; by 2007 total acres treated had reached 3.1 million acres[6]. At the county level, this reflects an average increase in acres treated as a percentage of acres planted from 7.1% in 2002 to 20% in 2007. This trend is not consistent with the predicted inverse relationship between GM adoption rates and insecticide usage.

The objectives of this paper are to: (1) investigate the increase in South Dakota acres treated with insecticide; (2) establish whether there is a link between the type of crop planted and acres treated with insecticide; (3) analyze the relationship between Bt corn adoption rates and corn insecticide usage in South Dakota; and (4) consider the larger implications of the study with respect to insecticide use in other states or regions with high GM corn adoption rates.

Literature Review

Analysis of chemical usage, specifically insecticides, is complex given the number of factors that influence usage, including advances in seed varieties, introduction of new chemical compounds and fluctuating pest infestation levels.

Recent studies have focused on various trends in pesticide use. For example, as new ground is broken for growing a crop there is a tendency for pesticide use to increase. The literature indicates that this occurred in the early 1980s. Osteen and Livingston[7] reported that there was an increase in planted acres and also an increase in pesticide application rates. The literature also discusses a recent trend toward mono-cropping or reduced use of rotations to meet market demands. This trend is in contrast to integrated pest management practices that reduce insect pest populations, and monocultures may even intensify pest pressure[8]. Indeed, biodiversity and species evenness are both critical aspects of agroecosystems' natural ability to resist pest pressure, and this ability is reduced or eliminated by monoculture cropping practices[9,10]. Insecticide use is thus necessary to control the resulting increasing pest populations in monocultures.

The presence of damaging insect pests is an important determinant of insecticide use. Intuitively one can assume that insect pest infestation levels are closely related to the amount of insecticide applied. That is to say, *ceteris paribus*, the more insects that are present the more a producer will spray. These infestation levels, however, can fluctuate from year to year, are often difficult to predict and can spread rapidly. Previous studies indicate that the inability to predict pest infestations can lead producers to spray insecticide prophylactically[11,12].

Insect pest infestation data have been used in previous studies to investigate the issue of producers making a simultaneous decision of insecticide use and GM adoption[13,14]. Perceived pest pressure likely affects whether a producer adopts GM seed and also how much they spray. By omitting pest pressure data there is likely a correlation between the error term and the dependent variable. Two-stage regressions are often used to model this type of decision-making process.

The most common empirical methodology used to investigate insecticide usage on agricultural crops is to evaluate changes in total pounds of insecticide applied[14–17]. In the literature, however, an evaluation of chemical usage (such as 'volume applied') ignores important chemical attributes like strength and application practices[7]. As a result, problems arise when trying to assess changes in insecticide application rates over time.

One alternative method that accounts for changing chemical attributes is a hedonic pricing model. A hedonic pricing model can provide a proxy for changes in chemical potency and breadth[14]. Another alternative empirical method proposed by Alston et al.[18] estimates chemical usage by multiplying the total acreage by the percentage treated for corn rootworm specifically. We have adopted a similar approach in this study. Our empirical methodology focuses on estimating the changes in chemical usage as a percentage of crop acres treated with insecticides in South Dakota.

There is a general consensus in the literature that total pesticide (herbicide plus insecticide) usage (total pounds applied) has increased due to increased GM acreage planted[19], largely driven by increased glyphosate usage associated with glyphosate-tolerant GM crops. The general consensus is GM adoption has decreased insecticide per acre usage on US corn acreage planted over the past decade[20]. However, Benbrook[19] argues that only a small proportion of corn acres were treated with insecticides prior to the introduction of Bt varieties. Therefore, the reduction in per acre insecticide usage associated with Bt adoption may be overestimated in the recent literature.

These findings reported by Benbrook suggest that the observed increase in acres treated with insecticide in South Dakota may indicate a reversal in the downward trend for insecticide usage by producers in states with high GM adoption rates. This trend reversal may be reinforced by unintended consequences associated with very high Bt adoption rates. For example, the introduction of Bt corn has allowed US producers to practice mono-cropping or reduced use of rotations to increase corn production. Insect populations facing greater exposure to a single environmental barrier may genetically adapt, thus reducing the effectiveness of the GM trait. Meihls et al.[21] reports western corn rootworm has the ability to adapt to Bt toxins if refuge acre policy recommendations are not followed, and Bt-resistant corn rootworms were recently discovered in Iowa cornfields[22]. Our goal is to make a contribution to the GM/pesticide debate by providing insight on increased insecticide usage in a state having a high GM adoption rate targeting corn insect pests.

Methodological Issues

Corn, soybean, hay and sunflowers are important crops for South Dakota agriculture and each has significant pest problems. Corn has several pests that cause crop damage every year, including the European corn borer (*Ostrinia nubilalis*) and the western and northern corn rootworms (*Diabrotica virgifera* and *D. barberi*, respectively). Soybeans in South Dakota have recently (2001) been invaded by the soybean aphid (*Aphis glycines* Matsamura) which is present in all soybean-producing regions of South Dakota (J.G. Lundgren, pers. comm., November 13, 2009). In South Dakota, 46 counties produced soybean in 2007. Additional information on soybean aphid infestation levels at the county level can be found at http://www.ncipmc.org/index.cfm. In 2002, the soybean aphid was present in 20 of 66 counties (30%) in South Dakota counties[23]. By 2007, the soybean aphid had spread to 46 counties in South Dakota ($\approx$ 70%). Alfalfa weevil (*Hypera postica*) affects alfalfa and is present in every county in South Dakota[24]. Sunflowers, because they are a native species and harbor an endemic insect fauna, face a substantial pest threat from 15 different insects[25]. Infestations can be highly variable even within a single sunflower field. Over time these pests have undoubtedly affected insecticide use. As this study does not directly link GM adoption to insecticide use, the inclusion of pest infestation data is not critical. However, we included a proxy variable for soybean aphid infestation at the county level in 2007. This proxy is potentially important because the scope of infestation and insecticide-use responses have changed dramatically over the time-frame of this study.

To estimate the effect of multiple explanatory variables on insecticide use for all counties in South Dakota, a fixed-effect model was employed. A fixed-effect model is often used on non-experimental data, where a scientific control group is not available or possible, treating each observation as its own control[26]. This model also accounts for time-invariant unobserved effects that are not captured with available data[27].

A typical ordinary least-squares (OLS) model often suffers from omitted variable bias. In the case of insecticide usage, the omitted variable is frequently the magnitude of the pest population. In South Dakota there are limited data on this particular issue, making it challenging to describe the relationship between insecticide use and other variables. Using the fixed-effect model we can assume there are two types of unobserved effects, those that vary over time and those that are constant, denoted as U_{it} and a_i, respectively[27]. The unobserved effect, a_i, accounts for unobserved effects that do not change over time (i.e., latitude, longitude, soil type and cultural norms). The time-varying error term U_{it} is subject to many assumptions that require some discussion. Examples include precipitation and technology adoption.

Accounting for pest populations is of particular importance. Some pest populations are unpredictable and can vary greatly from year to year. Also, pest communities have experienced dramatic change over the past 30 years. To account for the potential effect of soybean aphid pest populations at the county level we include a variable that identifies soybean aphid infestation presence in South Dakota in 2007. Other changes in pest populations are contained within the time-varying error term. We also make the assumption that pest populations are not correlated with crop choice as we believe there are other factors that have greater influence on crop selection, including market demands and prices as well as protective measures if pest infestations occur. If correlation between crop proportion and pest population does exist it would violate several of the assumptions necessary to ensure unbiased estimation[27]. Furthermore, we are not interested in how market factors are affecting cropping patterns. We consider this an issue for future research. Given the econometric caveats, our goal is to investigate how changes in cropping patterns may have affected the proportion of crop acreage applied with insecticide. Of special interest is corn acreage treated with insecticide because it is the only major crop in South Dakota that has a GM solution for insect pests.

Data

Data from several sources were used in this project. Data were collected from the National Agriculture Statistics Service (NASS) as well as the US Census of Agriculture. Data regarding total acres planted and crop-specific acres planted at the county level were retrieved from NASS's Quick 'Stats All States Data-Crops'[28]. The census data on chemical usage were collected from 'Agriculture Chemicals Used, specifically: sprays, dusts,

Table 1. Variables used in panel regression.

Variable name	Definition	Source/Description
%Treated	Dependent variable—percentage of total planted acres treated with insecticide	USDA Agriculture Census; 1978, 1982, 1987, 1992, 1997, 2002, 2007
YR07	Dummy variable for 2007	
Corn	Percentage of total acres planted with corn	USDA NASS
Soybean	Percentage of total acres planted with soybeans	USDA NASS
Sunflower	Percentage of total acres planted with sunflowers	USDA NASS
Hay	Percentage of total acres planted with hay	USDA NASS/ includes alfalfa
Aphid	Dummy variable for presence of soybean aphid in a county in a given year	Field survey by various extension educators (SDSU extension)
Intercept	Constant term	

Table 2. County-level summary statistics: percentage of acres planted: $N=462$.

Variable description	Mean	Standard deviation	Min.	Max.
%Treated	0.099	0.095	0	0.89
YR07	0.14	0.35	0	1.0
%Corn	0.22	0.16	0	0.64
%Soybean	0.12	0.15	0	0.51
%Sunflower	0.02	0.04	0	0.23
%Hay	0.28	0.19	0.035	1.9
%Aphid	0.14	0.35	0	1.0

granules, fumigants, etc. to control insects on hay and other crops'. The census reports all acres treated at the county level, but does not report crop-specific acres treated. The empirical hurdle we overcome is the lack of data on which crops are being treated, and by extension providing insight on why there has been an increase in acres treated as a proportion of acres planted at the county level. Soybean aphid infestation information by county is provided by the Plant Science Department at South Dakota State University and the Agricultural Research Service Station in Brookings South Dakota.

Empirical Model

The data set contains observations for each county over the past seven Census reporting periods (1978–2007). A fixed-effect modeling approach was selected to analyze the constructed panel data set. The data are a balanced panel, meaning that there are no missing observations for any counties over the observed years. Given the econometric issues discussed earlier, we used a 'cluster robust standard error' approach to account for both the intra-cluster correlation and between-cluster heteroscedasticity. Intra-cluster correlation can be caused by unobserved omitted variables (e.g., insect populations: see equation 21–13 in Cameron and Trivedi[29] for more details). A complete listing of the variables used in the model, and their definitions, is located in Table 1. Summary statistics for the variables are provided in Table 2.

The analysis uses South Dakota county-level data on acres planted with corn, soybean, sunflower and hay. In aggregate, these four crops accounted for 77% of all acres planted in 2007. The empirical analysis employs two regression models. The first regression model (Eqn 1) used all the crop variables as well as two dummy variables: (a) shift variable for 2007 and (b) a dummy variable to account for county-level aphid infestation in 2007. The second regression model (Eqn 2) includes interaction terms to capture the influence of each crop variable on insecticide usage in 2007.

The decision to include only the 2007 dummy variables (i.e., year, aphid) to capture changes in insecticide use over time is based on data indicating that insecticide use in South Dakota remained fairly stable from 1978 to 2002, with 8–10% of county-level planted acres being treated with insecticides. In 2007, South Dakota counties experienced an increase in acres treated with insecticides, on average; roughly 20% of the planted acres were treated with insecticides. This increase is depicted in Figure 2.

$$\begin{aligned} Y_{1\,it} = {} & \delta_1 + \delta_2 \mathrm{YR07}_{2\,it} + \delta_3 \mathrm{Aphid}_{3\,it} + \beta_1 \mathrm{Corn}_{1\,it} \\ & + \beta_2 \mathrm{Soybean}_{2\,it} + \beta_3 \mathrm{Sunflower}_{3\,it} \\ & + \beta_4 \mathrm{Hay}_{4\,it} + a_i + U_i t \end{aligned} \tag{1}$$

$$\begin{aligned} Y_{2\,it} = {} & \delta_1 + \beta_1 \mathrm{Corn}_{1\,it} + \beta_2 \mathrm{Soybean}_{2\,it} \\ & + \beta_3 \mathrm{Sunflower}_{3\,it} + \beta_4 \mathrm{Hay}_{4\,it} + \beta_5 \mathrm{Corn}^{*}07_{5i} \\ & + \beta_6 \mathrm{Soybean}^{*}07_{6i} + \beta_7 \mathrm{Sunflower}^{*}07_{7i} \\ & + \beta_8 \mathrm{Hay}^{*}07_{8i} + \alpha_i + U_{it}, \\ & \text{for } i \text{ county in year } t. \end{aligned} \tag{2}$$

The dependent variable in both regressions is a county-level variable and is defined as: $Y_{it} \equiv$ total acres treated with insecticide$_{it}$/total acres planted$_{it}$. By using the proportion of acres treated with insecticide as the dependent variable, instead of a weighted measurement of pesticides applied, we can avoid the problem of aggregating across a heterogeneous group of pesticides[30].

The outliers within the data identified in Figures 2 and 3 are worth discussion. Figure 2 shows the level and year of outlier occurrence, whereas Figure 3 provides

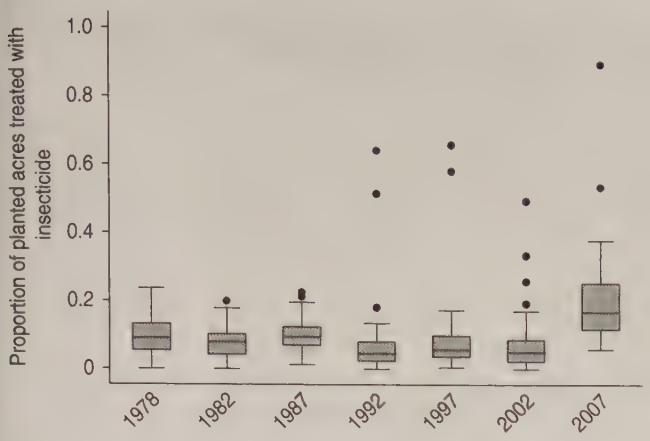

Figure 2. Proportion of planted acres treated with insecticide by county in South Dakota.

the geographical location of outlier counties where an unusually high percentage of acres treated with insecticides occurred. The outliers tend to form a cluster in the southeast region of the state, which is also a large corn and soybean production area. One 2007 outlier, Shannon county in western South Dakota, represents a county with very few planted acres, thus easily achieving a high percentage of their acres treated with insecticides. Using a robust model accounts for the outliers and therefore these observations were left in the data[31].

The crop variables are defined as a percentage of county-level total acres planted that are devoted to each specific crop for a specific county. County-level crop production patterns from 1978 to 2007 are provided in Figure 4. A change in a crop's acreage share at the county level is hypothesized to influence insecticide usage at the county level. Corn has recently experienced a large increase in acres planted at the county level, increasing from 23.5% in 2002 to 28.5% of total acres planted in 2007. Soybean acres planted have also dramatically increased over the span of the study, increasing from just 2.4% in 1978 to 21% in 2002 before dropping to 18% in 2007. Hay acres harvested and planted (1978–2007) ranged between 25 and 30% of acres planted at the county level. A majority of the hay acres harvested are located in western South Dakota. From 1978 to 2007, the proportion of acres planted with sunflowers was variable, ranging from 0.5 to 4% of county-level total acres planted.

The rapid expansion of the soybean aphid from 20 counties in 2001 to 46 counties in 2007 (all soybean-producing counties in South Dakota) has resulted in it becoming targeted as a major pest. To capture the aphid effect on insecticide usage a binary variable is included, set to one for counties when the soybean aphid is present in 2007 and zero otherwise. This variable is a proxy for aphid pest populations and it is hypothesized that the presence of the soybean aphid affected insecticide usage in South Dakota.

Results

Tables 3 and 4 provide coefficient and goodness-of-fit statistics for the two estimated models. The Hausman Test indicated that a fixed-effect approach is the appropriate choice for both models. The regression estimates for the first model indicate that corn, hay and aphid coefficients are statistically significant and are positively related to the proportion of acres treated with insecticide. The soybean and sunflower coefficients are not statistically significant. The estimated model has a global F statistic of 24.16, an R^2 of 0.35, and a correlation between model residuals and regressors of: $(u_i, Xb) = -0.3790$.

The initial fixed-effect model estimates indicated that the soybean parameter estimate was significant at the 1% level. However, when the model was adjusted for intra-cluster correlation the parameter estimate became statistically insignificant. This also occurred in the second regression, where the soybean parameter estimate was statistically significant at the 1% level prior to the regression procedure being adjusted for intra-cluster correlation. Therefore, we believe that our parameter estimates are valid but are insignificant due to multi-collinearity between the aphid, corn and soybean variables.

Table 3 indicates, *ceteris paribus*, that a 1% increase in corn acres planted will result in an estimated 0.24% increase in total acres treated in a county. The average proportion of county corn acres planted has increased from about 20% in the 1980s and 1990s to 28.5% in 2007. This suggests as corn acreage increases relative to alternative crops, insecticide usage increases. During this period South Dakota producers increased their planting of insect-resistant GM corn acres from 33% in 2000 to 59% in 2007, respectively (Fig. 1). The literature suggests that the relationship between GM corn and insecticide usage should be inversely related. That does not appear to be the case in South Dakota. These empirical findings may be reflecting the prophylactic application of neonicotinoid seed treatments[32] that often accompany GM trait packages in corn seed purchases. Or the increased usage of insecticide on corn acres planted may be due to producers abandoning traditional crop rotation practices, or breaking native ground to create additional crop land.

The statistically significant and positive coefficient for hay is also interesting. The hay coefficient implies that, holding the effects of the other variables constant, a 1% increase in the proportion of hay acres planted would result in a 0.27% increase in the proportion of acres treated. This may be the result of pest damage. Alfalfa, a subset of the hay variable, faces a significant pest threat from the alfalfa weevil. The alfalfa weevil has been present in South Dakota since 1973 and causes economic damage in many western South Dakota counties, where a large proportion of South Dakota alfalfa acres are planted[24].

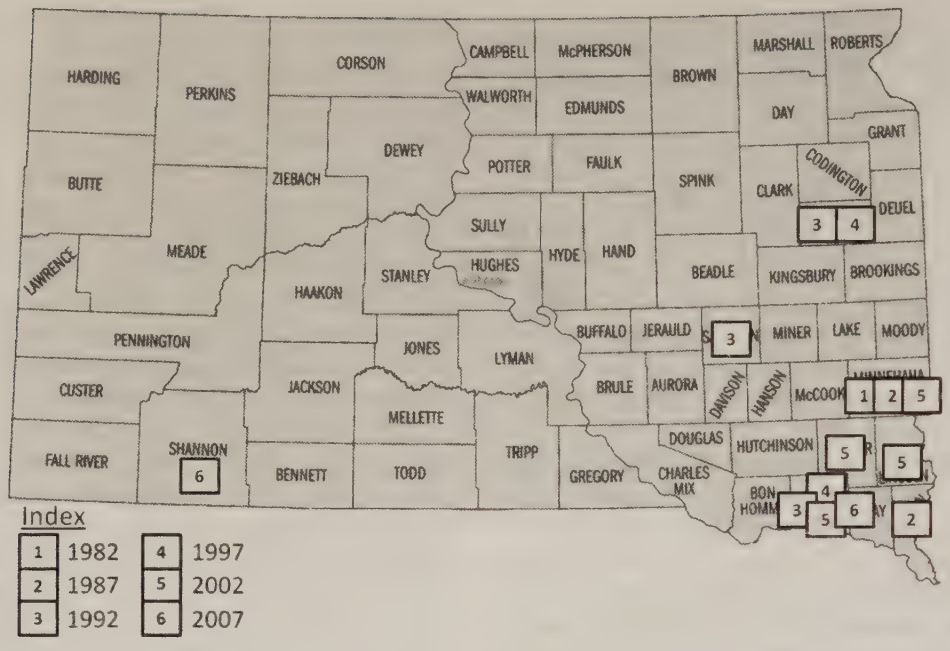

Figure 3. Map of South Dakota county outliers by year.

The soybean coefficient is statistically insignificant at the 10% level. As expected, the corn and soybean variables are highly correlated. The estimated correlation coefficient is 0.80. The aphid variable is also highly correlated with corn and soybeans, 0.40 and 0.45, respectively. Alternative estimation methods to deal with this high level of multi-collinearity were ineffective. Multi-collinearity inflates the standard errors but the coefficient estimates retain efficiency. The soybean coefficient is consistent with expected behavior. The only major insect pest threat to soybeans in South Dakota is the soybean aphid. Therefore, a negative soybean coefficient and a statistically significant positive aphid coefficient are consistent with the expected relationship between these variables and the dependent variable, i.e., the proportion of planted acres treated with insecticide.

The soybean aphid coefficient estimate is statistically significant at the 5% level. In 2007, *ceteris paribus*, it is estimated that counties with aphid infestation had 5.8% more acres treated with insecticide than counties without aphid infestations.

In Table 3, the 2007 year-dummy variable was statistically significant and positive. The regression coefficient indicates the estimated increase in total acres treated in 2007 is 6.1% higher than for the other census years in the study. This result corresponds to the noticeable increase in acres treated as shown in Figure 2. This statistically significant result prompted additional analysis that resulted in the inclusion of the 2007 interaction terms that are present in the second regression

The second regression and the inclusion of 2007 interaction terms is an attempt to gain insight on why

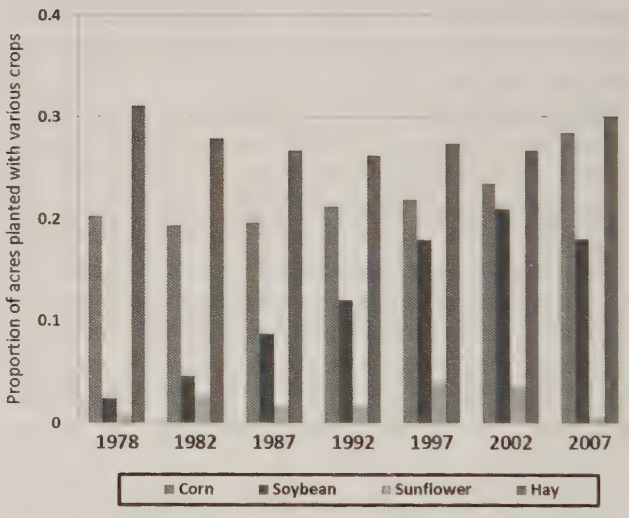

Figure 4. County-level crop acres planted in South Dakota.

acres treated increased in 2007. The interaction terms capture the change in the slope relationship between the percentage of acres treated and the proportion of acres planted for a specific crop for the year 2007. If the regression coefficient for an interaction term is statistically significant and positive, then the implication is that insecticide usage has intensified for that specific crop. The second regression model is statistically significant, with an F statistic of 51.27, an R^2 value of 0.36 and a correlation between model residuals and regressors of: $(u_i, Xb) = -0.2436$. The results for the second regression

Table 3. Results from first panel regression.

OBS = 462 No. of Groups = 66 OBS per group = 7	R^2: Within = 0.35 Between = 0.02 Overall = 0.10	$F(6, 65) = 24.16$ Prob > F = 0.00 Corr (U_i, Xb) = −0.379 RHO = 0.54 (intra-class corr)			
				95% CI	
Variable	Coefficient	Robust standard error	t-statistic[1]	Lower	Upper
Yr07	0.0612	0.0178	3.43***	0.025	0.097
Corn	0.239	0.080	2.99***	0.079	0.400
Soybean	−0.094	0.067	−1.39[2]	−0.230	0.041
Sunflower	0.0588	0.065	0.91	−0.070	0.188
Hay	0.271	0.093	2.92***	0.085	0.456
Aphid07	0.058	0.0269	2.17**	0.004	0.112
Intercept	−0.033	0.0314	−1.07	−0.096	0.029

[1] ***Significant at 1% level; **Significant at 5% level; *Significant at 10% level.
[2] The soybean coefficient has a P-value of 0.168.

Table 4. Results from panel regression analysis with interaction terms.

OBS = 462 No. of Groups = 66 OBS per group = 7	R^2: Within = 0.36 Between = 0.00 Overall = 0.16	$F(8, 65) = 51.27$ Prob > F = 0.00 Corr (U_i, Xb) = −0.243 RHO = 0.49 (intra-class corr)			
				95% CI	
Variable	Coefficient	Robust standard error	t-statistic[1]	Lower	Upper
Corn	0.2212	0.0862	2.57***	0.049	0.393
Soybean	−0.0974	0.0701	−1.39	−0.23	0.042
Sunflower	−0.0129	0.0602	−0.21	−0.13	0.107
Hay	0.2000	0.0880	2.27**	0.024	0.375
Corn*07	0.2309	0.1082	2.13**	0.014	0.447
Soybean*07	0.0231	0.1360	0.17	−0.25	0.295
Sunflower*07	0.4311	0.1242	3.47***	0.183	0.679
Hay*07	0.1054	0.0320	3.29***	0.041	0.169
Intercept	−0.008	0.0305	−0.27	−0.07	0.052

[1] ***Significant at 1% level; **Significant at 5% level; *Significant at 10% level.

are reported in Table 4. The 2007 dummy variables were dropped from the second model due to multi-collinearity issues.

The coefficient estimates for the crop variables were consistent with the first regression while the interaction terms lend some insight on the increase in insecticide use in 2007. The 2007 interaction variables for corn, sunflower and hay are significant and positive, implying an intensification of insecticide usage. The soybean and soybean interaction terms are statistically insignificant. The soybean parameter estimate was statistically significant at the 1% level prior to the regression procedure being adjusted for intra-cluster correlation.

Bt Corn and Insecticide Usage in South Dakota Counties

The regression estimates (Table 4) indicate that there is a statistical relationship between the proportion of corn acres planted and the proportion of acres treated. Holding all other variables constant the prediction equation is

$$E(Y_{2\,\mathrm{corn}}) = 0.2212^{*}\mathrm{Corn} + 0.2309^{*}\mathrm{Corn}^{*}07. \quad (3)$$

Equation 3 includes the corn*2007 interaction term. For all Census years except for 2007, the effect of the proportion of corn acres related to the proportion of acres treated with insecticide is given by the first term only. Thus, a 1% increase in acres planted with corn results in a

0.22% increase in the proportion of acres treated, *ceteris paribus*.

In 2002, on average for South Dakota counties, the proportion of corn acres to total acres planted is estimated at 0.2347. By substituting the average proportion of corn acres planted for 2002 into Equation 3, the predicted overall contribution of corn acres planted to the proportion of acres treated is estimated to be 0.0518 (0.2211*0.2347). The summary statistics for 2002 indicate that, on average, 7.1% of planted acres were treated with insecticide. Corn acres planted in 2002, on average, accounted for 73% (0.0518/0.071) of total acres treated and 24.6% (0.0518/0.2347) of corn acres planted in 2002 were treated with insecticide. In 2002, 43% of corn acres planted were Bt or stacked gene corn acres (state average). Therefore, Bt corn accounted for 0.1001 of the 0.2347 share of corn acres (0.43*0.2347) to total acres planted. Thus, the proportion of non-Bt corn acres planted accounted for 0.1346 of total acres planted. Assume Bt corn acres were not treated with insecticide (this assumption is made to in order to determine the upper bound of non-Bt corn acres treated). The implication is that in 2002, 38.5% (0.0518/0.1346) of non-Bt corn acres planted in South Dakota were treated with insecticide. It should be noted that the above estimates are based on the assumption that the percentage of acres planted with Bt seed at the state level is consistent with Bt plantings at the county level. County-level data on Bt usage is not available.

In 2007, on average for South Dakota counties, the proportion of corn acres to total acres planted is estimated at 0.284. The total effect of corn acres planted in 2007 on acres treated is determined by both coefficients in Equation 3. At the county level, the predicted contribution of the proportion of corn acres planted to the proportion of acres treated with insecticide is estimated to be 0.128, i.e. (0.2211*0.284 + 0.231*0.284). The estimated predicted value of acres treated associated with acres of corn planted in 2007 indicates that approximately 12.8% of the total acres planted were treated corn acres.

The summary statistics for 2007 indicate that on average, 20.1% of total (all crops) planted acres were treated with insecticide. Corn acres planted in 2007, on average, accounted for 63.8% (0.128/0.201) of acres treated. Furthermore, it is estimated that 45% (0.128/0.284) of corn acres planted in 2007 were treated with insecticide, a substantial increase relative to 2002 (24.6%). Given that 59% of corn acres planted in South Dakota contained Bt, our estimate for corn acres treated suggests that some Bt acres were treated in 2007.

These back of the envelope findings are surprising on a number of levels. First, GM corn acreage share and insecticide corn acreage coverage both increased. Secondly, in 2007, on average at the county level, corn accounted for 28.4% of total acres planted. Bt corn account for 59% of corn acres planted. Therefore, Bt corn accounted for 16.75% of total acres planted (0.59*0.284).

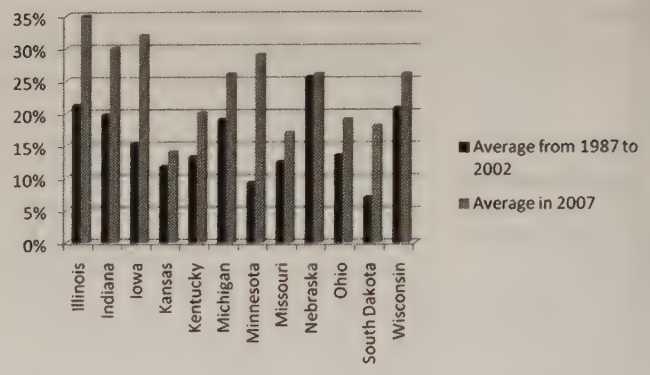

Figure 5. Proportion of acres treated with insecticide for mid-west states.

If we assume Bt corn acres were not treated with insecticide, then non-Bt corn acres accounted for 11.65% of total acres planted. The implication is that in 2007 109.87% of non-Bt corn acres planted in South Dakota was treated with insecticide (0.128/0.1165). We regard this estimate as an upper-bound for non-Bt acres treated and it suggests that South Dakota producers treated some Bt acres with insecticide in 2007.

This proportion is exceedingly high given: (1) that in 2002 it is estimated that 38.5% of non-Bt corn acres were treated acres; (2) traditional insect pest management patterns prior to GM introduction; and (3) the USDA refuge acreage requirement for producers who use Bt seed. Our estimate of a very large increase in corn acres treated raises some interesting questions. Has the change in cropping or seed marketing patterns forced producers to rely more heavily on chemical control? Is it the result of producers moving away from traditional rotation control methods used in the past to suppress insect pests? Are producers treating Bt corn acres in a manner they treat non-Bt acres? Additional research needs to be conducted on this issue.

Next, the coefficient estimates for the interaction terms found in Table 4 provide additional empirical evidence of increased acreage treated with insecticide being linked to changing cropping patterns. The corn interaction term coefficient estimate indicates that the slope relationship between corn acres planted and total acres treated with insecticide increased dramatically in 2007 from 0.2212 during the 1978–2002 period to 0.4521 in 2007 (0.2212 + 0.2309). This steepening of the slope indicates an increase in the marginal contribution of corn acres planted to total acres treated in 2007, relative to previous census years.

Overall, the empirical evidence suggests that corn, hay and sunflower production in South Dakota have experienced a recent intensification of insecticide use in 2007 relative to past Census reporting years. Empirical evidence indicates that the presence of the soybean aphid in South Dakota did contribute to increase the acres treated in 2007.

Broader Implications of the South Dakota Case Study

One question raised by the findings reported in this case study is whether the 2007 increase in acres treated with insecticide in South Dakota was an isolated phenomenon or whether other mid-western states experienced a similar increase. Figure 5 shows the proportion of acres treated with insecticide from 1987 to 2007. Of the 13 states that were looked at, Illinois, Indiana, Iowa, Minnesota and Kentucky seem to have experienced a marked increase in the proportion of acres treated in 2007. It is worth noting that while there appears to be a large and sudden increase, because of the limited nature of the data, it is unclear as to whether this increase was in fact gradual for the years in between 2002 and 2007.

For some states, like Illinois, the increase in the proportion of acres treated began in 1992. Other states, including Iowa, Minnesota, South Dakota, Indiana, Kentucky, had relatively low, though variable, proportions of their cropland treated with insecticides until the increase in 2007. The differences between states could be the result of varying or spreading pest populations, but more likely this is the result of a national paradigm shift in insecticide use in spite of the widespread adoption of Bt.

The 13 states listed in Figure 5 make up the Midwest region of the United States and are intensive crop-producing states. These states will provide the basis for further analysis at the county level and could provide greater insight into the reasons for the 2007 increase in the proportion of acres planted being treated with insecticide.

Conclusion

The empirical evidence gleaned from this South Dakota case study indicates that corn, hay and sunflower production in South Dakota has experienced a recent intensification of insecticide use. Furthermore, we have provided an empirical linkage to the increase in acres treated with insecticide in South Dakota to the intensification of insecticide use for specific crops. USDA data on acres treated with insecticide for other Midwestern states suggest that these states have experienced a similar recent increase in acres treated as is the case for South Dakota. The specter of a pattern of intensification of insecticide usage for crops coinciding with an intensification of GM crop production suggests additional research is needed.

Empirical evidence indicates that corn acres planted has a major impact on the proportion of acres treated with insecticide in South Dakota. This is partially due to corn being a major South Dakota crop. It is interesting that as GM adoption rose in South Dakota, so did the proportion of acres treated. There are a number of plausible explanations for this positive correlation. Producers may be treating more acres as an insurance policy against unnamed insect pests that are not targeted by Bt corn. The chemicals and products applied to crops today to protect against this and other non-target pests are often broad-spectrum products which will protect against a variety of potential threats, but also pose potential environmental harm to beneficial species[33].

Exogenous shocks affecting the market demand for corn may be affecting traditional crop rotation patterns, which have been a common cultural method for pest management practices. Producers may now be relying on insecticides (planting GM seed and seed and foliar applications) more heavily than traditional crop rotation to control pests, thus increasing the acres being treated.

As a final note, our study suggests that additional research is needed on the relationship between GM adoption rates and insecticide usage. Our analysis suggests that regions with high GM adoption rates may be experiencing unexpected insect pest issues not predicted in the previous literature.

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Introduction of domestic buffalo (*Bubalus bubalis*) into Ethiopia would be feasible

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Abstract

Due to its high agro-ecological variation and favorable production environment, Ethiopia has been endowed with high diversity of animal genetic resources and huge livestock population. However, the per capita consumption of livestock products is among the least in the world. To increase this consumption level, Ethiopia needs to keep livestock species that would be able to thrive under sub-optimal management systems, while providing some amount of products and services. The domestic buffalo (*Bubalus bubalis*) is one of the large ruminants that contribute to the least developed world's food and agriculture production. Therefore, introduction of the riverine-type buffalo would support food and agriculture production in Ethiopia. Moreover, the existing high demand for milk and meat, and the agro-ecological conditions are in favor of introduction of the riverine-type buffalo. More specifically, the riverine buffalo excels over indigenous cattle in its milk yield and quality of milk components. It also has a long productive life, a special ability of converting poor-quality roughage into animal protein, docile temperament and a wide range of adaptation. The presence of wild buffalo also shows Ethiopia's suitability for domestic buffalo rearing. Therefore, even though it seems illogical to make direct comparison between the performance of the Ethiopian indigenous cattle and the domestic buffaloes kept elsewhere; the hardy nature of buffalo and its wide range of adaptation would indicate the adaptation of the riverine-type buffalo in Ethiopia. However, the introduced riverine-type buffalo needs to be handled on experimental stations until management expertise is developed.

Key words: domestic buffalo, introduction, Ethiopia, feasibility, indigenous cattle

Introduction

Ethiopia is one of the eight Vavilovian centers of biodiversity in the world. As a result, it has been well known for its high diversity of animal genetic resources and huge livestock population. According to the Domestic Animal Genetic Resources Information System (DAGRIS)[1], Ethiopia is a home for 32 recognized cattle breeds, 7 sheep breeds, 13 goat breeds and 10 chicken ecotypes. Moreover, there are four donkey breeds, two horse breeds, two mule types and four camel breeds[2]. This is despite the paucity of phenotypic characterization work especially for livestock breeds that are kept by minority ethnic groups and pastoralists inhabiting the very remote areas.

The livestock sector in Ethiopia accounts for 13–16% of the national GDP and 30–35% of the agricultural GDP and constitutes 15% of the total export revenue[3]. This contribution has been made by 43.1 million cattle, 23.6 million sheep, 18.6 million goats, 6.5 million equines (donkeys, horses and mules), 0.62 million camels and 34.2 million chickens mainly kept by subsistence smallholder farmers in the highlands and the pastoralists in the lowland areas[4]. This huge livestock population has placed Ethiopia among the high-ranking countries in Africa and for some of the species at the international level. More specifically, Ethiopia is known to be first in its cattle population and second for its sheep (next to South Africa) and goat population (following Nigeria) in Africa. Ethiopia also constitutes 39.63% of the African equine population[5]. Accordingly, its donkey population is the second in the world following China, and its camel population is the third in the world (next to Somalia and Sudan). Therefore, this indicates that Ethiopia has suitable agro-ecological conditions for rearing different types of livestock species.

This huge livestock population and high genetic diversity have ecological, historical, socio-cultural and economic perspectives. For example, an ancient country, Ethiopia has a long-lasting tradition of farming, it is characterized by

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high agro-ecological diversity and, according to Taddese[6], it constitutes 70% of the African highland mass (>1500 m asl), which is considered to be a suitable environment for farming and human settlement. Accordingly, it was estimated that about two-thirds of the total land mass is suitable for agricultural activities[7]. Moreover, diversified uses of livestock species and their multiple socio-economic roles are among the main driving forces behind this diversity. Ethiopia is also the homeland for more than 80 ethnic groups that have at least some degree of variation in their practice of livestock farming. This ethnic diversity could be a source of animal genetic resource diversity, because farmers and pastoralists of a given ethnic group might have specific selective breeding practices dictating their needs.

However, the per capita consumption of livestock products in Ethiopia is among the least in the world. For example, according to FAO[5], the per capita consumption of milk was 22.5 kg and of meat was 8.5 kg. The main reasons for this low level of consumption are low amount of products obtained from indigenous livestock due to production constraints such as feed shortage, disease problems etc., together with the inherently low genetic potential of indigenous breeds. It was also noted that recurrent drought, poor infrastructure and market development, weak policy supports and institutional arrangements have reduced the performance of the livestock industry[7]. Moreover, to feed the fast-growing human population (around 3% annual growth), a considerable portion of the grazing land has been increasingly converted into cropland. This has forced smallholder farmers to rely upon highly fibrous and poor-quality crop residues such as straw, stover and stubble, and marginalized grazing lands to feed their cattle. This type of feeding regime was not able to support, for example, the nutritional requirement of improved temperate cattle breeds and their crosses (attempts have been made for decades to improve the performance of indigenous cattle through cross-breeding with temperate breeds) and as a result these improved breeds usually produce below the expected level[8].

However, Ethiopia has a favorable environment for rearing different types of livestock species, especially those already acclimatized to sub-optimal management and low-input–low-output farming systems. The domestic buffalo is one of these species that have indispensable importance in the least developed countries (LDC) for food and agriculture production. Therefore, the objective of this analysis is to synthesize the existing evidence on domestic buffalo production systems and to analyze the feasibility of introducing the riverine-type buffalo into subsistence-oriented Ethiopian farming systems.

Ethiopia's Agro-Ecological Diversification

Ethiopia is located in east Africa and is bounded by geographical coordinates 3°24′–14°53′N and 32°42′–48°12′E. It has a total area of 1.1 million km^2 and, of this, an area of 7444 km^2 is occupied by major lakes. Its altitude ranges from 125 m below sea level in the Denakil depression to 4543 m asl on Mount Ras Dashen[9]. Average annual rainfall varies from < 100 mm in the arid eastern and south-eastern regions to 2400 mm in the southwest highlands, with a national average of 744 mm[10]. This wide variation in altitude and rainfall pattern has resulted in 18 major agro-ecological zones that can be further divided into 49 sub-zones[11]. Ethiopia has major rivers that are 7000 km long, including the famous Blue Nile River that contributes ~85% of the Nile's water source. Ethiopia also shows wide variation in its temperature, which ranges from below 0°C to more than 40°C depending on the altitude and season of the year. However, in the agriculturally important areas, the average daily temperature usually ranges from 10 to 30°C. This is an ideal temperature for buffalo production, because 20°C is the optimal environmental temperature for rearing of buffaloes[12] and this falls well in the temperature range of agriculturally suitable areas in Ethiopia. However, the best milk breeds of buffaloes (the riverine-type) in India and Pakistan are mostly confined to areas where the summer temperature rises above 46°C and the winter temperature falls below 4°C[13].

The Domestic Buffalo

The domestic buffalo (*Bubalus bubalis*) is one of 40 livestock species used in world food and agriculture production. The domestic buffalo species can be divided into two sub-species or types; the riverine- and swamp-type[14–16]. The diploid set of chromosomes ($2n$) in the riverine-type is 50, whereas it is 48 for the swamp-type[17]. The swamp-type is mainly used for draft and secondarily for meat[14,15,18], therefore, it is considered as a dual-purpose animal[14]; whereas the primary purpose of keeping riverine-type buffaloes is for milk and secondarily for meat[18]. However, the riverine-type was noted by McCool[14] as triple-purpose animal (milk, draught and meat). Therefore, in both types meat production has secondary importance. Similarly, Ranjhan[19] reported the primary use of buffalo for milk and secondarily for meat production in South Asia. However, Sadana[20] reported the use of swamp buffalo to produce small amount of milk and in line with McCool[14]; Sadana[20] has also reported the use of the riverine-type for transport. The swamp buffalo is the most common type in typical South-East Asian rural villages[14]. However, for example, in India, the riverine-type is widespread throughout the country[20]. Generally, the riverine-type is found in South Asia, whereas the swamp-type is distributed across South-East Asia[16]. Swamp buffaloes are typically smaller and less productive compared to their riverine-type counterparts, whereas the riverine-type is characterized by high milk yield because for decades it has been selected for this trait[15].

Common buffalo breeds

India and Pakistan are the home of the best milk breeds of riverine-type buffalo in the world, Murrah and Nili-Ravi;

whereas swamp buffaloes are not distinguished into breeds[20]. Therefore, breed description and diversity in India are made for riverine-type buffaloes only. Moreover, by virtue of its number and importance, the domestic buffalo in India is synonymous with the riverine-type; however, the majority of buffaloes in Vietnam, Thailand, China and adjoining countries are of the swamp-type[20]. Important buffalo breeds also include the Jaffarabadi and Surti[21]. Jaffarabadi is commonly used in many countries for cross-breeding due to its large size and to exploit its high carcass yield potential[18]. However, the popular Murrah is known as 'black gold' in India and is widespread throughout Asia and the rest of the world[20]. However, Singh and Barwal[22] reported the superiority of the Nili-Ravi breed of Pakistan over Indian Murrah and Egyptian buffaloes. Buffalo breeds can also be classified based on their horn orientation. For example, horns are closed and set close to head and are down swept in Murrah, Nili-Ravi, Mehasana, Jaffarabadi and Sambalpur, whereas horns are sickle-shaped and non-swept in Bhadawari, Kalahandi, Kanara, Manda, Nagpuri, Pandharpuri, Tarai and Toda[22].

Population trend

The world's buffalo population is estimated at 170 million head and approximately 97% are found in Asia, whereas the remaining 2% and 0.3% are found in Africa and Europe, respectively[23]. In Europe and Asia, there is a growing interest in the use of buffaloes for milk and meat production. As a result, the riverine buffaloes have been introduced into many European countries and in swamp buffalo-dominated areas[15]. Murrah buffalo has been the breed of choice for introduction and is followed by Nili-Ravi, Jaffarabadi and Surti. This is because among domestic buffalo breeds Murrah is a well-known milch breed[20]. The importance of buffalo in South Asia is increasing faster than that of cattle, although in some East and South-East Asian countries buffalo population has declined rapidly[19]. This is due to the massive use of farm mechanization which has significantly reduced the swamp buffalo population[18]. Despite this, in the majority of the south Asian countries, a major shift has been taking place from cattle to buffalo dairying. The reasons for this include consumers' preferences for higher butterfat content, persistency of milk production under poor management systems and longer productive life of buffalo[21]. Due to this, for example, buffalo population in the Brazilian Amazon increased at a rate of nearly 13% per annum from 1975 to 2000 compared to cattle with an annual growth rate of 4% only. Therefore, buffalo has steadily gained acceptance as an alternative to cattle, and throughout South America buffalo performs better than cattle both in meat and milk production[24].

Behavior

Buffaloes are probably among the easiest mammals to train and are more rustic animals compared to cattle[15]. Therefore, buffalo is considered a cumbersome and docile large ruminant[14]. Hence buffalo's rusticity and its ability to adapt to different environments are among its important characters[25]. Buffaloes are therefore generally pretty good in their temperament; however, like most animals, the more they are handled the better is their temperament[26]. Buffaloes have inherent docility, therefore they can easily learn upon training and the occurrence of deliberate attacks on people is perhaps much rarer in buffaloes compared to other livestock species[27].

Ecological adaptation

Buffalo rearing has gained importance in recent years due to its innate ability to survive in adverse agro-ecological conditions and due to its special ability of converting poor-quality roughage and crop residues into animal protein[28]. As a result, buffalo rearing is an integral part of the traditional smallholder mixed farming system[25]. Buffaloes also have a special ability to subsist on coarse feeds, straws and crop residues and to convert these feed materials into protein-rich lean meat[29]. Moreover, buffaloes are well known for their disease resistance and special ability to survive and thrive under hot, humid climates[12,18]. This has made buffaloes well recognized for their better adaptation to poor feeding, harsh climate and infectious diseases[30]. Moreover, buffalo is found in more than 50 countries of the world, varying in their ecology, climate, topography and socio-economic conditions. Buffalo adapts well from tropical conditions of the Indo-Pakistan region to the temperate climate of Italy[30]. Therefore, this wide range of adaptation of domestic buffalo[13] indicates its ability to acclimatize to Ethiopia's environment characterized by high agro-ecological diversity. Cattle suffer badly while transiting from dry to wet seasons; however, these conditions bring few problems to buffalo. Due to its ability to adopt and even thrive under difficult conditions of the floodplains, buffalo has now become a popular choice among floodplain ranchers[24]. Moreover, due to long-term natural and artificial selection, buffaloes possess many biological features, including resistance to high temperature and humidity[31]. However, compared to cattle, thermo-regulation is poor in buffalo, therefore, often it requires shade or water for wallowing, although the riverine-type that is proposed for introduction into Ethiopia has better thermo-regulation than the swamp-type[13].

Utility Pattern

Buffalo is a triple-purpose animal that provides milk, meat and draught[19]. Moreover, buffalo has resistance to common bovine diseases, shows superior weight gain than cattle, produces high-quality milk and meat products and is able to fatten on a wide range of pastures[24]. Buffaloes produce milk that is rich in nutrients and high in quality and provide draft power and dung/urine as organic fertilizer and provide meat that is known for its flavor and marbling, and other products such as mozzarella cheese[20]. Buffaloes contribute

72 million tonnes of milk and 3 million tonnes of meat annually to world food production and this is mainly produced by smallholder farmers[22]. For example, buffaloes in India account for 30% of the total meat production[19]. Buffaloes represent 12% of the world bovine stock, but its share in the world bovine meat export was estimated at 27% in 2007[32]. However, some buffalo products have a lower price compared to cattle products. For example, buffalo meat is sold at about half the price of beef and buffalo hide is sold at ~40% of cattle hide. The live buffalo is sold at 14% of cattle price. The low export value of buffalo meat could be mainly due to low world market demand for buffalo meat and low export value of buffalo hide could be due to low quality and demand. The low price of live buffalo could be mainly due to export of weaned calves rather than fed bulls or steers and/or low carcass weight of the exported buffalo. Despite the low price of exported buffalo meat, India exports by far more buffalo meat than beef[32]. This might be due to the worship of cows by Hindus, which, in turn, could affect the practice of cattle slaughtering. Generally, buffalo meat is less expensive than cattle beef[24] and is much cheaper than chevon, mutton, pork and poultry. Therefore, it is a cheap source of animal protein for the part of the society with weak purchasing power[19].

However, in the case of milk, the market demand and price show a different picture. For example, in Italy, keeping buffaloes is generally considered more remunerative than keeping cattle. This is because the price of buffalo milk is 2.3–2.5 times more than that of cow's milk and this is about 2.5 times in Asia[33]. Similarly in Pakistan, buffalo milk is sold at Rs. 20 per liter, compared to cow's milk which is sold at Rs. 15[34]. Moreover, Payne and Wilson[13] reported the production of excellent, thick and tough leather articles from buffalo hide, hence leather could be considered an important by-product derived from slaughtered and fallen buffaloes[19]. Moreover, on average, about 5 tonnes of dung are produced annually from an adult buffalo[19]. An adult buffalo therefore voids 3–5 kg dry matter and 50–100 g nitrogen every day, besides potassium and other minerals of high manure value[35]. However, 8.4% and 9.3% of the fuel source in the urban and rural areas of Ethiopia, respectively, constitutes cow dung. Therefore, this type of cow dung use competes directly with the use of dung to improve the soil nutrient levels[36]. The massive horn of buffalo is also used to produce fancy and decorative horn articles[13].

Milk production

Under rural subsistence farming systems the average milk yield of buffaloes is 1350 liters[37]. Approximately 10.75% of the total milk yield is produced in the first month of lactation, whereas ~12.5% and 11.25% were produced in the second and the third month of lactation, respectively[22]. In Italy, buffalo cows are commonly milked twice a day with an interval of 11–12 h[38]. Buffalo's milk is traditionally preferred over cow's milk by the majority of Pakistanis[37,39,40] due to its aesthetic and creamy characteristics, high milk fat (6–8%), flavor and slightly higher solid not fat (SNF) content (~9–10.5%). Due to this, buffaloes account for 62% of the total milk production in Pakistan[40]. Moreover, the ratio of butter or ghee output from buffalo milk was approximately three times more than that from cow's milk. Buffalo milk is also highly preferred in soft cheese making and the ratio of milk to soft cheese is 5 : 1 kg compared to 8 : 1 for cow's milk[37]. Moreover, Nili-Ravi and Murrah breeds on average complete three lactations in their production lifetime[22], which shows their prolific nature.

Milk fat percentage of buffaloes usually exceeds cattle by 1–3%, depending on the breed type and environmental condition and compared to cow's milk, buffalo milk has a higher percentage of all milk components[41]. The average fat globule size of buffalo milk is larger and its viscosity is higher (2.04) compared to cow's milk (1.86). Due to its higher dry matter content, buffalo milk has a higher cheese yield, however, this does not seem to be due to the difference in milk quality alone. Moreover, for each protein percent point more than 5 cheese yield points are obtained for buffalo, whereas it is about 3 for cows[41]. Buffalo milk also has low cholesterol and high calcium content compared to cow's milk[37]. Moreover, despite its high fat percentage, milk and mozzarella cholesterol content are lower in buffalo's milk than cow's milk (275 mg versus 330 mg and 1562 mg versus 2287 mg, respectively)[41]. Buffalo milk also contains high levels of the natural antioxidant tocopherol peroxidase, having 2–4 times the activity of cow's milk. Furthermore, there are a growing number of people who suffer from cow's milk allergy but this is not the case for buffalo milk[37].

However, according to the findings of Mukasa et al.[42] in the central highlands of Ethiopia lactation milk yield of indigenous cattle was 238 ± 82 liters with associated lactation length of 239 ± 55 days. Payne and Wilson[13] reported that milk of short-horned Zebus in Ethiopia has a fat percentage ranging from 4.7 to 7.1. Therefore, for the majority Ethiopians who prefer to consume a large amount of ghee by adding it as a flavor in a sauce that is consumed together with the traditional bread called '*enjera*', the high-fat percentage of buffalo milk is readily acceptable. A high cheese yield from buffalo milk would be highly demanded by Ethiopians because cheese is among the commonly consumed milk products. Buffaloes are, however, slow and hard milkers due to their slow milk ejection reflex and their hard teat sphincter muscle, although the pressure during milking is higher in buffaloes than in cattle[38].

Meat production and weight gain

Buffaloes, used as draught animals for centuries, have well-built muscles. However, until recently, little attention was given to their use exclusively for meat production[29]. Despite this, due to its low fat and calorie content, buffalo

meat has high demand in India and on the international market[28]. This is also due to the lean, tender and juicy nature of buffalo meat[19]. Buffalo meat is also the healthiest among red meats due to its low calorie and cholesterol content. It has about a 2–3-fold cost advantage over mutton and goat meat. Furthermore, its use in meat processing is on the increase due to its high lean meat content and low fat percentage[29]. Therefore, due to buffalo's lean nature[29]; most of the buffalo meat from India was exported for meat processing industries[32]. However, about 86% of the world buffalo meat is produced in Asia and is mainly obtained from old and culled animals. Meat from these types of buffaloes is dark, less tender, has a strong odor and hence it is poorly accepted by consumers[22]. This is indeed true, because meat is mainly produced when buffaloes are retired from their productive lives[19]. However, when young buffaloes are raised under an intensive feeding system their meat is lean, tender and highly palatable and is favorably compared with beef from cattle of similar age and weight[22]. The subcutaneous fat layer of the buffalo carcass is usually thinner than that of comparably fed cattle. Thus, the buffalo carcass has a rounder rib, higher proportion of muscle and lower proportion of bone and fat compared with beef. More pigmentation or less intra-muscular fat (1–2% marbling compared with 3–4% in beef) content makes the buffalo meat darker[29]. However, the dressing percentage of buffaloes is slightly lower than that of cattle[22]; and old and unproductive buffaloes have 43–57% dressing percentage[29]. However, this is higher than the dressing percentage of 42–53% reported for Small East African Zebus[13].

Growth is linear in buffalo from birth up to 36 months. The average daily gain ranges from 548 g (3–6 months) to 404 g (birth to 36 months)[22]. However, the Murrah calves grow at a rate of 900–1000 g a day with feed conversion ratio of 5 : 1[19]. It was also found that buffalo calves gained more and consumed less amount of feed per unit of gain under restricted feeding systems compared with cow calves[30]. Therefore, buffaloes have excellent body weight gain compared with cattle[29].

Ethiopia is a home for wild buffaloes and there had been a tradition of hunting wild buffalo in some areas as a symbol of heroism and to consume bush meat. However, this tradition has been gradually eroded due to change in the socio-economic, cultural and political atmospheres. Therefore, this shows that the consumption of buffalo meat will not face any restrictive social taboos in Ethiopia.

Draught work

Buffaloes are a major source of draught power, and that is why they have been called live tractors of East and South-East Asia[14,22]. This is because buffaloes provide steady but sure-footed draft power in the remote rural areas[20]. Therefore, the draught work of buffalo is among its most important uses in the least developed world[14]. For example, in India 60% of the total farm power is generated from draught animals, of which about 10% is contributed by buffaloes. A buffalo can pull loads of more than six times its body weight, but its common load carrying capacity is 1.5–2.0 tonnes, i.e., 3–4 times its body weight. It can pull this load continuously for 2–3 h and for 6–8 h a day during winter and 5–6 h a day during summer with rest in between[22]. Similarly, it was reported that the usual load carrying capacity of buffalo was 1.5–2 tonnes continuously for 3–4 h during summer and 6–8 h a day during winter[19]. The swamp-type buffalo is especially important for work on paddy fields[20]. Therefore, even though buffaloes are slow movers compared to cattle, they can pull heavier loads[19].

Management Practices

Health care

Although mastitis is not a common problem in buffalo species, it is a costly disease; therefore, the exposure of teat ends to pathogens should be minimized[38]. However, the hardy nature of buffaloes minimizes disease problems and allows the use of simpler and cheaper housing and equipment[33]. In buffaloes, the incidence of hemorrhagic septicemia and internal parasites were the highest, followed by foot and mouth disease and pneumonia. Other diseases/disorders include tympani, mastitis, dystocia, rabies and lameness[34]. Major infectious diseases that cause significant losses in Bangladeshi buffalo, for example, include foot and mouth, rabies, contagious bovine pleuropneumonia, hemorrhagic septicemia, anthrax, black quarter, bovine tuberculosis, brucellosis, calf scour and calf diphtheria. Brucellosis is the main cause of abortion in buffalo[21]. Therefore, a comparable type of health care practiced by smallholder farmers and the veterinary services in Ethiopia can be adopted to control the risk of disease problems in the riverine buffaloes.

Feeds and feeding

Buffaloes are reared in a mixed cut and carry, tethering and grazing on communal pasturelands and using crop residues and agricultural by-products[18]. Buffaloes also feed on crop leftovers[20] and they can perform better on roughage and marginalized poor pastures[15]. As a result, buffaloes have developed the special ability of converting poor-quality roughage into milk, meat and other services and products[20]. The superiority of domestic buffalo over cattle in digesting and efficient use of feeds is clearly observed when the two species are kept on a low level of nutrition[30]. It was also found that buffalo calves digest larger amounts of dry matter and nitrogen-free extract than cow calves under restricted feeding systems[30]. One of the characteristics that make buffaloes preferable is therefore their extraordinary ability to convert fibrous feed into energy, and this enables them not to compete with humans for scarce resources such as grains[25]. Due to its size, a buffalo's rumen harbors large microbial populations that facilitate better digestion of feed.

Rumen acidosis is also seldom found in buffalo due to a high rate of saliva secretion which helps buffalo to maintain its rumen pH[30]. Therefore, for countries such as Ethiopia that commonly rely upon poor-quality roughage such as straw, stover and stubble to feed their cattle, the high efficiency of domestic buffalo in digesting poor-quality roughage is a reward for resource-poor smallholder farmers.

Reproduction

Buffalo has low inherent reproductive capacity, resulting in late sexual maturity in heifers followed by longer inter-calving period and fewer calf crops[39]. Domestic buffaloes are generally known for their poor reproductive efficiency. However, this is largely due to the production environment under which the majority of them are maintained, such as harsh climate, poor nutrition and sub-optimal production inputs. Therefore, they would have good fertility if managed well and properly fed[16]. Buffaloes reach sexual maturity at later ages; however, they have extended longevity. As a result, buffalo cows are well grown before their first calving, and this can help them to maintain their body condition while nursing their calf. This extended longevity ensures that buffalo cows produce more calves in their productive lifetime[14]. Therefore, a low replacement rate is required due to their long reproductive life[33]. Moreover, although buffaloes attain puberty at a later age compared to cattle, they have a remunerative longer reproductive lifetime[16].

The average age at first calving (AFC) and calving interval (CI) were 48 and 20 months, respectively, under smallholder buffalo production systems[33]. Specifically, the average calving interval in Surti buffaloes was 462 days[22]. Moreover, according to Younas et al.[25], the average age at maturity and CI were 34.8 and 17.5 months, respectively. Buffalo heifers usually attain puberty when they reach approximately 55–60% of their adult weight, but the age at which this is achieved largely depends on breed, nutrition, management, climate and year (season) of birth[16]. The gestation length of buffaloes is 312–334 days and is longer than that of cattle (285 days)[26]. These reported findings on AFC and CI of domestic buffaloes are comparable with the reproductive performance of indigenous cattle under smallholder farmers' management systems in Ethiopia.

Buffaloes come into heat regularly throughout the year; however, the highest is in October and the lowest is in April[22]. Similarly, buffalo cows and bulls are capable of breeding throughout the year but they often show seasonal variations in fertility due to climatic and nutritional factors that modulate ovarian and testicular activities[16]. Buffaloes usually come into heat in cold months and are sub-fertile during hot months[22]. In spite of this, buffaloes are poly-estrus and are capable of breeding throughout the year[16]. Natural mating is still the most common breeding practice in buffaloes. The use of artificial insemination (AI) both in the swamp and the riverine buffaloes is still marginal and is less efficient. This is due to small herd size, poor heat detection and poor management. This is because buffalo's reproductive function is largely affected by environmental factors and it has physiological features that make it more similar to that of sheep than cattle[15].

Table 1. Crude protein (CP) percent content of commonly used crop residues in India and Ethiopia.

Feed type	Ethiopia[45]	India[46]
Rice straw	–	4.0
Wheat straw	3.9	3.5
Teff straw	5.2	–
Finger millet straw	–	3.7
Maize stover	–	4.6
Sorghum stover	–	4.2
Barley straw	4.7	–
Natural grass hay	6.6	–
Faba bean residue	7.2	–
Field peas residue	6.7	–

Table 2. Climatic and agro-ecological setting of India and Ethiopia.

Variable	India[47]	Ethiopia
Average temperature (°C)	12.5–30	16–30[48]
Average annual rainfall (mm)	1000–1500	744[10]
Average humidity (%)	49.2	–
Elevation (range)	0–8598 m asl	125 m bsl–4543 m asl[9]

m asl, meters above sea level; m bsl, meters below sea level.

Production Systems of Riverine Buffalo in its Breeding Tract (Mainly India) versus Indigenous Cattle in Ethiopia

An attempt was made to compare the production systems of the riverine buffalo maintained in LDC (with special emphasis on India) with production systems maintaining indigenous cattle in Ethiopia. This comparison is made mainly on feed resource base and climatic and agro-ecological indices. The riverine-type buffalo is mainly kept by resource-poor smallholder farmers in the LDC and the same is true for indigenous cattle management in Ethiopia. Therefore, both the riverine buffalo of the LDC and the indigenous cattle of Ethiopia have evolved under fairly comparable types of management (feeding, health care and housing) systems.

The feed resource base

Buffaloes have subsisted for centuries on low-quality fibrous feeds such as rice and wheat straw, which are low in their protein and energy content, and while grazing

on natural pasturelands[43]. Moreover, in South-East Asia, buffaloes are mainly fed on cereal straws that are highly lignified and are low both in their fermentable protein and carbohydrate contents[44]. The most limiting nutrient (crude protein, CP) content of these fibrous feeds used in buffalo feed in India is comparable with the commonly used crop residues to feed cattle in Ethiopia (Table 1). It was also noted that majority of the Asian buffaloes are reared in a mixed cut and carry, tethering and grazing system on communal pastureland[44]; which also reflects the feeding systems practiced in Ethiopia. Therefore, this showed that the riverine buffalo can readily acclimatize to the feeding regime of Ethiopia's smallholder farmers who practice a fairly comparable type of feeding system.

Climatic condition and agro-ecological setting of India and Ethiopia

The climatic indices in Ethiopia fall in the range of the extremes of India's climate (Table 2). Therefore the riverine buffalo can adapt to the climatic conditions of Ethiopia as it did in the Indian sub-continent. Moreover, distribution of the riverine buffalo and Zebu cattle usually overlaps in India. Thus, as Zebu has already adapted to the conditions in Ethiopia for millennia following its introduction from its presumed center of domestication (around India), the riverine buffalo is also expected to have a similar type of adaptation in Ethiopia. Ethiopia's agriculture, with the exception of the non-sedentary pastoral areas, is entirely based on a mixed type of crop–livestock farming[49]. This shows that crop production and livestock farming have co-existed as complementary farming activities for millennia[49,50]; however, competitive interaction is observed when the commonly used natural resources, such as land, are limited[50]. As indigenous cattle have been part of this mixed crop–livestock system, it is expected that, unlike deforestation and the practices of inappropriate tillage that are noted as the main causes of soil degradation[51], the proposed introduction of riverine buffalo would fit in this farming system as do cattle, with a relatively small negative effect on the natural resource base. However, improving the grazing land management and the balanced use of crop residues as a source of soil nutrient and feeds would have to be practiced to reduce the depletion of soil nutrients arising from the use of crop residues to feed buffalo and to minimize the outflow of nutrients from the soil[50]. This can be further supported by planting improved pastures and by the use of alternative energy sources such as biogas. This is because the slurry produced from biogas plants can be used as a nutrient source to improve the fertility of the soil. These types of management practices are quite important to make the rearing of riverine buffalo a sustainable farming activity. This is indeed true because sustainable agriculture, among others, must be based on the development of appropriate types of soil management practices[52].

Performance and Adaptation of Swamp versus Riverine Buffaloes

The swamp buffalo has narrow range of adaptation compared to the riverine buffalo and, due to its very nature, it usually requires unlimited access to water to keep itself cool and is lighter in weight. The riverine buffalo, on the other hand, is heavier and its milk yield is much higher[13]. Therefore, for Ethiopia, where paddy farming is an unimportant practice, introduction of the riverine buffalo would have comparative advantages over the swamp buffalo to increase milk and meat production of the domestic herbivores.

Conclusions and Recommendations

The diverse agro-ecological zone of Ethiopia and the wide range of adaptation of domestic buffalo showed the feasibility of introducing riverine-type buffalo into Ethiopia. Moreover, the riverine buffalo is hardy, has higher milk yield and fat content compared to indigenous cattle and, more importantly, it can thrive well on low-quality roughage. Therefore, introduction of the riverine-type buffalo has economic advantages besides introducing an additional agriculturally important livestock species. However, the reproductive behavior of buffalo is less intense than that of cattle, therefore, understanding this behavior requires thorough follow-up when buffalo cows are in heat. Moreover, maximum care should be taken before introducing the riverine-type buffalo. Thus, identification of suitable areas to establish experimental farms and training of local experts on husbandry practices by experts from countries that have extensive experience of riverine buffalo rearing are required, as recommended by Taye and Ayalew[53].

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The role of green roof technology in urban agriculture

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Abstract

Urban agriculture is a global and growing pursuit that can contribute to economic development, job creation, food security and community building. It can, however, be limited by competition for space with other forms of urban development, a lack of formalized land use rights and health hazards related to food contamination. The use of green roof technology in urban agriculture has the potential to alleviate some of these problems, without adversely affecting the benefits provided by urban agriculture. It would not only enable the use of land for development and agriculture, but may also facilitate the formation of formal space and water use agreements, and enable redistribution of ground-level resources among urban farmers. This could decrease the use of contaminated land and water at ground level and alleviate health concerns. Before green roof technology can be incorporated into urban agriculture on a larger scale, installation costs must be reduced, roof weight limitations should be assessed, and appropriate management practices should be developed which will ensure that the benefits of green roofs, such as energy savings and storm water management, are still provided to urban communities.

Key words: Eco-roof, local food production, vegetated roof, vegetables, community garden

Introduction

In recent years, the importance of green space in urban areas has been changing. A new vision of urban centers incorporates more green space, such as parks[1], and a more human-friendly environment mixing traditional urban centers of industry, commerce and residence with food production[2,3]. Implementation of this vision has been facilitated by the introduction of more environmentally friendly technologies and the introduction of policies and programs which promote their use[4–8]. One such technology being incorporated into development is green roofing.

Definition of green roofs

Green roofing is a technology enabling growth of vegetation on rooftops, effectively replacing green space lost during building construction. Conventional green roofs generally consist of a number of layers including a root barrier to prevent damage to the underlying structure; a drainage layer to facilitate the removal of excess water; a filter fabric to prevent the drainage layer from clogging with media; growing media; and vegetation[9]. Others may be composed of growing modules or vegetated mats. Green roofs vary in depth of growing media and vegetation, but are generally divided into two categories: extensive (those with <15 cm of media) and intensive (those with >15 cm of media)[7]. Extensive roofs are usually planted with ground cover or succulent species that require little maintenance after establishment, while intensive roofs can support herbaceous perennials, shrubs and even trees[9,10] but typically require continued inputs. Design and plant selection depend on the purpose of the project and the environmental benefits to be achieved. Uses of green roofs range from functional storm water and energy management roofs to park-like amenities open to the public, to food production.

Factors determining plant survival include media composition and depth[11,12], incoming solar radiation[13], climate and most importantly soil moisture[14–16]. Insufficient moisture can be remedied by altering media composition and depth or with irrigation if available. Growing media used on commercial green roofs are often engineered and

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comprise mostly lightweight materials such as heat-expanded slate or shale[9]. Media composition depends on manufacturer and available materials, but it is designed to be lightweight while maintaining the ability to support plant life. Organic matter content may vary and is beneficial for plant growth, but when it decomposes it may leach nutrients, resulting in runoff water-quality issues, and is inconvenient to replace. Economics dictates that substrate composition will depend on materials that are locally available and can be formulated for the intended plant selection, climatic zone and anticipated level of maintenance.

Benefits of green roofs

Use of green roofs in urban development has been shown to provide a number of benefits including reduced air and noise pollution, carbon sequestration, increased habitat and biodiversity, increased roof lifespan, storm water retention, energy savings and mitigation of the urban heat island[9,13,17–26]. Computer modeling predicts that an increase in green roof area, as could take place with the technology's incorporation into urban agriculture, will amplify these benefits[27]. Moreover, the introduction of green roofs to city development plans could promote infill development (the redevelopment of vacant areas within urban centers) and reduce spending on the development of new infrastructure such as roads and sewer lines[22]. Although infill development largely benefits municipalities and companies, expansion of the green roof industry would increase employment and economic growth in urban centers, benefiting populations that typically turn to urban agriculture.

Energy savings[28–31] and mitigation of the urban heat island[17,27,31–34] in particular have great practical benefit to poorer communities who are most likely to benefit from urban agriculture. Green roofs can reduce energy consumption of a building by 2 to 39%[29,30] and by between 12 and 87% for the top floor[30] during the summer. These reductions are primarily due to reducing the amount of direct solar radiation that reaches the roof and the amount of heat transferred into the building[28,31]. The extent of the reduction is dependent on the extent of vegetative cover on the roof[26,28,29], the existence of other insulation[30], the thickness of the green roof media, irrigation of the roof and climate[29]. Energy savings during the winter are negligible in warm climates, but can be seen in cooler climates[29,30]. Energy savings in very humid climates are also expected to be lower due to reduced evapotranspiration and the associated cooling effects[17].

Green roofs have been shown to reduce ambient temperatures[31], an effect that is projected to increase with increasing roof surfaces and the inclusion of green walls (plants grown on walls using a variety of training and planting systems) in urban development[17,27]. Modeling predicts a corresponding reduction in urban temperatures and increase in thermal comfort[17,27]. The end result of reduced temperatures and therefore energy use is a reduction in the use of conventional air-conditioning[17,30]. This would benefit low-income neighborhoods for two reasons. Urban neighborhoods with higher summer temperatures than surrounding urban areas are correlated with lower incomes, higher poverty rates[32] and impoverished individuals aged 65 and older[33] in cities such as Phoenix, AZ and Philadelphia, PA. These individuals are more at risk to environmental hazards, such as extreme heat events, due to a lack of resources enabling them to cope with the hazards, such as air-conditioning[32]. Implementation of green roofs would not only reduce energy bills in such neighborhoods but also reduce the occurrence of extreme heat and the need to air-condition in the summer time, freeing up limited funds for other uses and improving the quality of life of individuals in these at-risk areas and populations.

Figure 1. Vegetable production on a green roof in East Lansing, MI.

Food production in an urban setting

Food production can be added to the benefits provided by green roofs[22,35] (Fig. 1) and expanded through the incorpoation of this technology into urban agriculture. Urban agriculture is defined as horticultural, agricultural or farming activities carried out on small plots of land in and around urban centers[36–38]. Individuals in urban centers around the world participate in urban agriculture for reasons, such as poverty, unemployment, food insecurity[2,39], high prices of market food, income or asset diversification and supplementary employment[2]. These motivational factors[2,3,36–41]; limitations such as land availability, land use and ownership rights, physical and economic access to inputs and potential food contamination[36–39,42,43]; and geographic and climatic factors unique to each urban center shape urban agriculture[2,3,36,38]. It is likely that a number of these limitations could be alleviated through the use of green roofs, while maintaining the benefits expected by urban farmers. Currently, policy is under reform in a number of cities worldwide resulting in accommodation for urban agriculture[3]. These policy changes present an opportunity and could be guided to include farming on green roofs and expedite the inclusion of green roofs and other alternative

technologies in urban farming with great benefit to urban populations.

Despite the possible benefits from incorporating green roof technology into urban agriculture, there are a number of potential issues that must be addressed. These include installation and maintenance costs, weight limitations, media composition and depth, cultural practices, potential water-quality issues of effluent and how food production would influence the other known benefits attributed to green roofs. These are not only factors that may limit the use of green roofs, but would also limit their viability for widespread use in urban agriculture. Further research and innovation may present solutions to these problems. The goal of this review is to examine some of the possible benefits and barriers to incorporating green roof technology into urban agriculture. In doing so, we shall first examine the current state of urban agriculture and then introduce potential benefits or limitations of using green roof technology. Due to the lack of published studies on the subject, we will also discuss future research needs. We have therefore divided the review into sections containing common themes; economic benefits and food security, economic barriers, access to resources and policy, human health and environmental health concerns.

Economic Improvement and Food Security

Economic development brought about by participation in urban agriculture comes in a variety of forms, including the supplementation of family income, job creation and freeing up funds previously used to purchase food. Crop choice and scale affect the extent to which urban agriculture contributes to the income of a household[2,37–39]. Rice, for example, is a staple in many parts of the world and can provide income security for an urban farmer's household[38], but production of vegetables may yield higher market prices[37,38]. Animal husbandry, another form of urban agriculture, can provide high profits[2,37,38], but may require much higher investments[38]. In some cases, social capital can be generated by a household, by giving away food that could not be sold[2]. The impact of urban agriculture on employment is highly variable, depending on the economic status of urban farmers[2,37]. For households in both developing and developed countries that do not produce for sale, or sell only their excess produce, urban agriculture frees up funds for other uses[2,36,38,39]. The prevalence of urban agriculture increases when poverty increases and when costs of purchasing food surpass that of growing it[2]. This can be an important measure in stretching household budgets, allowing for the purchase of other items[2,39] or some economic freedom for women where household budgets are male-controlled, as was found in Pretoria, South Africa[39]. In addition, economic concerns are an incentive for consumers who assume that purchasing local produce increases economic returns to local farmers through shortened supply chains and better market accessibility[40,41].

Figure 2. A community garden in Detroit, MI.

Food security, the second major driver of urban agriculture, is affected by both quantity and quality of food available to a household. Even in locations where urban agriculture does not contribute significantly to employment, food security is of major concern to urban farmers[2]. Currently, food in the US must be shipped an average of 74 km (49 miles) to consumers in urban centers[41]. It has been estimated that under ideal conditions the agricultural products of the entire state of New York could not supply the agricultural needs of more than 55% of New York City[41]. This suggests that providing adequate food supplies for urban centers with growing populations may not even be possible on a regional scale, and costs associated with shipping may affect the food security of the urban poor. Producing agricultural goods within urban centers is one method of reducing the ecological footprint of urban centers[3,41] and ensuring urban dwellers, access to food. Food insecurity, or the lack of access to adequate food for an active and healthy life[17], is not just a problem in the developing world, but in the US as well[2,36,44] (Fig. 2). Food insecurity can be temporary or chronic[3] and is associated with a variety of problems in adolescents, who are at higher risk than young children[44]. A perceived or actual need to improve food security and a lack of ability to rely on food from rural areas can result in the use of urban agriculture[3,37], which has been shown to improve the quantity and

Figure 3. A herb garden on a green roof in Grand Rapids, MI.

quality of food available to low-income urban households under a variety of conditions[2,3,36,37,44].

Green roofs are already utilized to improve the economic circumstances and food security of urban farmers (Fig. 3). EcoHouse, in St. Petersburg, Russia is an example of a rooftop garden project which provides jobs to, and increases cash flow among, individuals living within the apartment complex[35]. The project also provides those residents with a reliable source of vegetables[35]. Another example is the green roof of the Fairmount Hotel in Vancouver, a portion of which is devoted to a kitchen garden, saving the hotel approximately $30,000 a year[22]. The rooftop garden on Earth Pledge's New York office is used not only as a source of food but also as a promotional tool for the group's organic local produce campaigns[4,22]. Similar community garden projects have been developed in other cities, such as the Multnomah County Green Roof Project in Portland, OR[45] and several community-scale gardens in Chicago, IL[46]. A green roof at Trent University in Peterborough, Ontario, is producing vegetables for a local restaurant, The Seasoned Spoon Café, which was started as a healthy fast-food alternative[47].

Economic Barriers

Economic barriers to urban agriculture include inadequate access to or knowledge about markets[2] and insufficient labor[2,27], inputs such as fertilizers[38], quality seeds[37] and credit or subsidy for startup costs or inputs[2,36–38]. These barriers are due to limited resources of urban farmers and will not be affected by the introduction of green roof technology to urban agriculture. An additional barrier introduced by the use of green roof technology is the cost of green roof installation and maintenance.

Installation of a green roof can be $32 m^{-2} more expensive than a conventional roof for roof structure alone[26]. Installation of green roof systems can vary from two to six times more expensive than conventional roof systems, depending on the design of the roof system[26]. Factors that impact the cost of a green roof include ease of access for installation, structural integrity of the building, type of drainage system, depth and composition of media, inclusion of an irrigation system and the use of a modular, mat or conventional built-up continuous roof system (C. L. Rugh, personal communication, February 5, 2010). Maintenance costs of a green roof also depend on roof design, as intensive roofs tend to require more care than extensive roofs. Maintenance of the roofing layers themselves is comparable to that of a conventional roof due to the longer lifespan of green roofs[26].

Tapping into incentive programmes, such as those used in Portland, OR[6] and Chicago, IL[5] which provide reductions in storm water removal fees and grants to help subsidize installation, may help. Other programmes, such as those geared toward improving the availability of fresh fruits and vegetables in urban centers and improving healthy eating habits in urban youth may also be sources of funding. It is also possible that locally made materials could be used to construct green roofs, reducing the cost of installation, but generation of policy promoting the use of green roofs and subsidizing their installation will be of greater importance in low-income areas than in high-income areas. Evaluation of locally available materials would also be necessary to determine both their suitability and their impact on green roof installation costs.

Access to Resources and Policy

Availability of land, especially land of adequate quality, is the main obstacle affecting urban agriculture[2,3]. Land scarcity and uncertainty in maintaining access to available land are due to competition with other development uses, primarily building construction[2,37,38]. These other uses are often more economically profitable and are therefore preferred by land owners[3,39,43]. Land use and investments in urban agriculture by urban farmers are impacted by resource use rights. Under current systems there are often no formal leasing agreements between land owners and urban farmers cultivating vacant lots[2]. Rights of urban farmers are often minimal[3], uncertain[43], and frequently transient due to changing land uses and termination of informal use agreements[39,43]. Lack of formal agreements over water use rights has led to conflict between municipalities and urban farmers[39].

Incorporation of green roofs into new development would increase the potential agricultural area and remove competition with urban development that reduces the willingness of urban farmers to invest in urban agriculture[2,38]. Currently, flat rooftops comprise as much as 85% of the roof area in downtown and commercial areas[20]. In larger urban areas, this could add up to a great deal of space and potential for productive use if these existing roofs were retrofitted into green roofs. Buildings must, however, have the structural integrity to support the added weight in a worst-case scenario, regardless of whether roofs are designed new or retrofitted[48]. Some existing roofs may not be suitable for retrofitting without incurring considerable costs in structural support. Many flat roofs have load capacities of only 146 $kg\,m^{-2}$ (30 ft^{-2}), which could be exceeded by as little as 7.6 cm (3 inches) of growing media[49]. This means that the flat roof area of existing buildings, which could be used for urban agriculture, is not accurately represented by the flat roof area of a city. More information on which roofs can support the additional weight of a green roof and the minimum depth of media necessary for agricultural production will enable more accurate estimates of how much roof area could be used. Despite these limitations, land owners could take advantage of this potential, enabling them to utilize more profitable development and then generate secondary profits through rental agreements with urban farmers.

Development of flat roof space into agriculturally productive areas could facilitate formalization and standardization of rental, leasing or use agreements between land owners and urban farmers. Access to roof space is limited and would require urban farmers to negotiate with building owners to gain access to the green roof space. This would be a reversal of the current use of vacant lots for urban agriculture, whose absent owners are unaware of agricultural activities or unwilling to take measures to keep urban farmers off the land[2,3]. Although this could create problems for urban farmers if green roof owners are unwilling to rent the space due to zoning or building code issues that might arise[50,51], it could also empower urban farmers. Formal, legally binding use arrangements would grant urban farmers recourse should the green roof owner break the agreement. Such formal and empowering leasing agreements could also encourage farmers to increase investments in urban agriculture, increasing productivity and food security. Urban farmers with informal arrangements do not currently have this level of power and security[3,39,43]. Formalized rental and leasing agreements could easily be extended to include access to the buildings, water supply. This would be greatly beneficial to those farmers who have expressed willingness to pay for clean water where no clean water source currently exists[37], but will increase the costs of farming for most urban farmers. Rainwater collected from an unused portion of the roof may also provide an added source of clean water for irrigation.

There are, however, two additional potential outcomes of rental agreements for the use of green roof space. First is the exclusion from farming and water resources of resource-poor urban farmers unable to pay rent for green roof space. Second is the reallocation of ground-level space and water sources currently used by farmers able to pay for rental agreements. The former could result in greater problems associated with poverty and food security in urban areas, but the latter could grant a larger number of urban dwellers access to land and water and therefore the economic opportunities and additional food security provided by urban agriculture. If the latter is the outcome, it would mean a better quality of life for a greater number of urban dwellers. This will be of particular importance as populations become increasingly urban, both worldwide and in the USA[39,52,53], and doubts about the ability of rural areas to agriculturally support these growing urban populations also increase[39,41,44].

Human Health Concerns

Access to fertilizers is especially important as space limitations in urban agriculture require more intensive farming and greater fertilizer use per area than rural areas[36]. Often resource-poor urban farmers will use inexpensive and easily accessible fertilizers, such as manures or municipal wastes, which can lead to an increase in soil heavy metal and pathogen concentrations[36,37,42,54,55]. Heavy metal and pathogen contamination of food are the primary human health concerns associated with urban agriculture. Sources of contamination include soils in which crops are grown, water used for irrigation and air pollutants. In many cases, the land most readily available to urban farmers is contaminated with heavy metals from a variety of industrial and mining sources, which can lead to contamination of the agricultural products[40,42,54]. In addition, resource-poor urban farmers often cannot afford to pay for clean irrigation water even if a source exists[2,37,38]. Atmospheric deposition of contaminants during production, transportation and marketing of produce also leads to elevated levels of heavy metals[40,55–57].

The extent to which heavy metal contamination affects the safety of vegetables depends on several different factors, including the vegetable species, the part of the plant that is eaten and the type of heavy metal[54,55,57,58]. On average, fruit vegetables accumulate lower quantities of heavy metals than leafy or root vegetables[54,57,58]. Leafy vegetables are a major source of heavy metal dietary intake[55], due to high rates of translocation, transpiration and growth, as well as high surface area in close proximity to contaminated soil and irrigation splash[54]. High-surface-area vegetables, such as cauliflower[56], and those that spend more time in the field[57], are also known to accumulate higher concentrations of heavy metals through atmospheric deposition. Dietary intake of heavy metals can lead to accumulation in the human body because they are non-biodegradable[59], causing their negative effects to become apparent only after years of exposure[55,59]. Among resulting health problems are a variety of cognitive disruptions[59,60], nervous, cardiovascular[54,56,60], kidney, bone and liver[54,56] diseases, as well as cancer[59].

The use of green roofs in urban agriculture also has the potential to reduce health concerns. Green roofs have the potential to reduce use of contaminated land in urban agriculture due to the nature of their construction. In most cases, the media in which vegetables would be grown on green roofs are engineered instead of using local soils, and so initial contamination will be minimal. Green roof media are also less likely to accumulate heavy metals than ground soils due to their high permeability and low cation exchange capacity, which results in leaching of nutrients[61] and heavy metals[62] into runoff water. This tendency for leaching would reduce the likelihood of vegetable contamination on green roofs if contaminated sources of water or fertilizers are used. Rental and water use agreements will also facilitate more widespread use of uncontaminated water sources for irrigation. Atmospheric deposition of contaminants may also be reduced during the production phase. It has been suggested that distance from the source of pollutants impacts the extent of heavy metal contamination due to atmospheric deposition[56]. Most green roofs are several stories high, increasing the distance between crop production and such sources of pollution as major roadways and highways.

In addition to food contamination, concerns about urban agriculture include health problems due to improper handling of agrochemicals and urban waste, a potential increase in pests such as rodents and flies, which can contribute to the spread of diseases and the transmission of diseases from livestock to humans due to improper animal husbandry techniques[3]. Although these concerns may be avoidable through proper practices, they promote negative perceptions of urban agriculture. Formalized leasing agreements may provide greater oversight of agrochemicals and urban waste used in urban agriculture. Leasing agreements could include specifications for what, if any, agrochemicals or urban wastes can be used on the green roof, how they should be stored and used, and the consequences for the urban farmer if the specifications are not followed resulting in human injury or a health hazard. In some countries, organizations may already be in place which could monitor such agreements, such as the Occupational Safety and Health Administration (OSHA) in the USA. It is unlikely that the use of green roofs in urban agriculture will affect the keeping of large livestock, which would be impractical on rooftops.

Environmental Health Concerns

Despite the costs, urban farmers are currently producing vegetables on green roofs in natural soils and composts at a media depth of 30 cm or more. This practice potentially creates several problems, including the added weight to the roof, consistency of growing media, potential nutrient loads polluting effluent that discharges into our waterways from fertilizers and as compost decomposes, and the logistical practicality of adding compost every year on a roof several stories above the ground.

Water quality of runoff is another concern as nutrient leaching could cause problems downstream. Composition of the growing media is one aspect of this problem. Most commercial green roof media are formulated within the guidelines of the German FLL (Forschungsgesellschaft Landschaftsentwicklung Landschaftsbau) standards[63]. Years of experience have resulted in media that possess the chemical and physical properties to support plants, yet are lightweight, coarse enough to allow drainage at shallow depths, can be replicated and limit nutrient runoff when guidelines are followed. Natural soil is always variable and the addition of compost can lead to runoff quality issues. Even so, commercial green roof media generally contain about 15–20% organic matter which can result in nutrient loading[19,23,64]. Researchers in North Carolina[64] found that concentrations of both N and P decreased with decreasing percentages of compost in the media. These results emphasize the point that growing media can have an immense effect on the quality of effluent. In addition, applications of fertilizers and pesticides to ensure plant growth can be very detrimental to water quality[23,61]. This is especially true for soluble fertilizers applied in liquid form[61].

Studies performed on green roofs with conventional media have shown that nutrient leaching can initially be a problem, but overall, these media can have a positive effect on water quality[23,65]. The initial nutrient load likely is due to decomposition of organic matter that was incorporated into the original mix. Once established, and the organic matter reaches an equilibrium, vegetation and substrates can improve water quality of runoff by absorbing and filtering pollutants[23]. The effect of plants and their root systems was evident when effluent from an unplanted green roof containing media was shown to have higher concentrations and totals of N and P than effluent from planted roofs[65].

There is also the question of how much fertilizer is necessary to maintain agricultural productivity in green roof media. Experimentation has shown that non-succulents

grown on green roofs require either additional organic matter in the media or fertilization[12]. The relatively high levels of fertilizer that may be necessary to produce vegetables could lead to high levels of nutrient leaching. This begs the question, are we trading the benefits of local food production for decreased water quality? If nutrient loading does turn out to be a problem, then green roofs could be coupled with other low-impact development practices such as rain gardens and bioswales (landscaping techniques designed to manage storm water), although these practices are not always possible in dense urban settings. This highlights the need to develop and use green roof growing media and cultural practices that minimize leaching of nutrients while still providing adequate physical and chemical properties for plant growth.

Conclusions

In addition to previously mentioned research needs, there are several areas where research on the use of green roof technology in urban agriculture is necessary before wide-scale use of the technology can be implemented. First, determination of what crops are suited to growth in green roof media will be necessary. Little is known about how growing vegetables on green roofs will impact the environmental benefits provided by green roofs. Many of the benefits are directly related to the amount of coverage achieved by the vegetation and the leaf area of the vegetation[66]. The coverage that would be achieved by vegetables will be very different than that of the ground covers and perennials traditionally used on green roofs because they are typically cultivated in rows. In addition, vegetable gardens would be replanted every year, whereas typical green roofs are populated with perennial species. Research on how this difference will impact energy savings and storm water retention, for example, will enable better assessment of this use of green roofs in areas where these benefits are of particular importance.

The effects of other environmental factors on crops, such as exposure to higher winds, should be determined for optimum crop selection. Although pollinators have been seen and kept on green roofs, an understanding of the efficiency and quality of pollination of vegetable plants on green roofs would enable better decision-making about which crops to grow and the necessity of bee keeping on vegetable-growing green roofs. Finally, economic evaluation of different crops may generate more information on how much economic impact this form of food production could have on both a small and large scale.

The incorporation of green roof technology into urban agriculture maintains the economic and food security benefits of urban agriculture, while eliminating some of the many difficulties faced by urban farmers around the world. The ideal case, where formalized use agreements with building owners and oversight by municipal authorities ensure greater space availability and healthier produce, is, however, only possible through the cooperation of all parties involved, something that may be difficult in areas where urban agriculture is viewed in a particularly negative light. The formalization of use rights required by use of green roof space by urban farmers represents an opportunity for farmers to achieve guaranteed access to quality land and irrigation water, providing security for their agricultural pursuits. For land and building owners, the formalization of use rights represents an opportunity to achieve greater economic success and some degree of oversight over the activities taking place. This combination of economic opportunity and oversight may have the added benefit of improving land and building owners' attitudes toward urban agriculture, which could expedite policy reform. Municipal involvement will enable new insights into the benefits of urban agriculture and understanding the ways in which its negative impacts can be minimized.

The process, though difficult, could be made easier through the establishment of policy friendlier to urban agriculture, incentive and subsidy programmes for the installation of green roofs and research into reducing the initial cost of green roofs and minimizing the inputs necessary for productive agriculture on green roofs. The resolution of these issues will further enable a future in which urban areas are greener and healthier places to live. This future could utilize ideas about development that incorporate green space in the forms of green roofs, parks or agricultural plots, enabling a closer connection with nature and the production of food with the benefit of increased food security, especially for the urban poor.

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A European perspective for developing modern multifunctional agroforestry systems for sustainable intensification

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Abstract

One of the key questions of primary importance to global agriculture and food security is how to optimize sustainable intensification to balance competing demands on land for food and energy production, while ensuring the provision of ecosystem services and maintaining or increasing yields. Integrating trees and agriculture through agroforestry has been attracting increasing interest as an agroecological approach to sustainable intensification. Trees have traditionally been important elements of temperate agricultural systems around the world, but there has been increasing separation of agriculture, forestry and nature over the past few decades. This paper discusses what we can learn from traditional agroforestry systems to help develop modern systems that integrate ecological farming and agroecological advances to achieve sustainable intensification. We also discuss the existing barriers to wider adoption of agroforestry, and identify how these barriers can be overcome to promote agroforestry as a mainstream land-use system.

Key words: agroecology, food security, ecosystem services, pollarding, orchards

Introduction

One of the key questions of primary importance to global agriculture and food security is how to optimize sustainable intensification to balance competing demands on land for food and energy production while ensuring the provision of ecosystem services and maintaining or increasing yields[1]. Agroforestry has been identified by the International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD) as a 'win–win' multifunctional land use approach that balances the production of commodities (food, feed, fuel, fiber, etc.) with non-commodity outputs such as environmental protection and cultural and landscape amenities[2]. Agroforestry land-use systems integrate trees with crops and/or livestock, with ecological and economic interactions between the trees and other components[3]. Evidence is now coming to light that shows agroforestry systems can deliver a range of ecosystem services and environmental benefits, including biodiversity conservation, regulation of soil, air and water quality, and carbon sequestration[4,5].

Trees have traditionally been important elements of temperate agricultural systems around the world, evolving from systems of shifting cultivation toward more settled systems involving agriculture, woodland grazing and silvopasture, with fertility transfer from woodlands to cultivated land via manure[6,7]. These early systems integrating trees and agricultural production began to decline during the Middle Ages, when crop rotations were developed to maintain soil fertility rather than relying on transfer of nutrients from woodlands to croplands. This separation was further enhanced by the introduction of chemical fertilizers during the 19th century and the development of separate policy regimes for forestry, agriculture and nature conservation over the past 60 years[6]. This paper discusses what we can learn from these traditional agroforestry systems to help develop modern systems that integrate ecological farming and agroecological advances to achieve sustainable intensification[8]. We also discuss the existing barriers to wider adoption of agroforestry, and identify how these barriers can be overcome to promote agroforestry as a mainstream land-use system.

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Traditional Agroforestry Systems

Fruit tree systems

Up until the 20th century, fruit and nut silvoarable (trees with crops), and silvopastoral (trees with pasture/livestock) systems covered large areas in central Europe. These systems are still widespread in certain countries; in Sicily almond trees with cereals or fodder cover 18,000 ha and fig trees with cereals cover over 10,200 ha in Crete and the Aegean islands[6]. However, in large parts of Europe, other traditional agroforestry fruit tree systems have been declining. The *pomeradas*, apple trees planted in lines or scattered throughout meadows and croplands, in humid areas of northern Spain have existed since the 13th century but have been declining dramatically over the past 35 years[6]. In southern Europe, mixed vineyards that incorporate trees as mechanical support for the grapevines, with the added bonus of diversified economic return, have also greatly declined due to intensification and increased mechanization[6].

Olives

Dating back to pre-Roman times, when rows of olive trees were intercropped with wheat in alternate years to improve the olive yield the following year, olive groves still cover an estimated 650,000 ha in Greece and 20,000 ha in Italy[6]. Grown in rows or as scattered trees, the olive trees are intercropped with cereals, vegetables and fodder crops, and systems combining olives with grape vines are still found in Spain (46,600 ha) and Greece[6].

Shelterwoods and woodland grazing

The practice of pasturing in woodland is one of the oldest land-use practices in human history. In northern Europe, mature woodland provided shelter to cattle and sheep during the winter months, while in Mediterranean regions woodland provided browse, forage and shade for livestock during early summer drought periods, and in turn grazing controlled the understorey, reducing the fire risk[9,10]. Developed over centuries, transhumance systems (extensive animal husbandry involving seasonal relocation of livestock) are still found in mountain areas throughout Europe from Norway to Switzerland, Italy and Spain[11–13].

Wood–pasture remnants in England feature some of the oldest and widest trees in Europe, providing valuable resources for a wide range of associated biodiversity, as well as having historical and cultural value[14,15]. The New Forest in southern England is one of the largest remaining areas of wood–pasture in temperate Europe, with over 3000 ha of woodland grazed by ponies, deer, cattle and pigs. Recently formed into a National Park, the New Forest pasture woodland has high biological and cultural value and must be grazed to maintain its unique nature.

Pollards

Pollarding, cutting branches from trees 2–3 m above ground, is an ancient practice that provided fodder for livestock and/or wood for fuel or other uses. Archaeological excavations have uncovered pollards dating back to the Iron Age[16], and a fossil oak pollard found during gravel extraction in the UK has been carbon dated as 3400 years old[15]. Fodder pollards were often established in wooded meadows where a hay crop was cut from under the trees and grazed subsequently, while wood pollards were more widely found in wood pasture where grazing occurs for most of the year[17].

Pollarding for fodder was practiced across Europe, and was particularly common in northern Europe and mountainous areas such as the Pyrenees, Alps and high pasture areas of the Basque country[17]. Leaf-bearing branches from pollards were cut on a 2–6-year rotation in the summer, dried and stored for use as livestock feed during the winter. Leaves were also plucked by hand from the trees and dried as a livestock fodder, particularly for dairy cows, and fallen leaves raked and collected in the autumn, primarily for bedding[16]. Fodder tree systems are still found in the Mediterranean—deciduous oak leaves are shredded and dried for sheep fodder in Greece, while in Crete and Sicily, carob pods are stored for fodder[6]. Pollarding for wood was also widespread across Europe, though often concentrated in specific areas to meet special demands, such as in Epping Forest, England, where industrial-scale pollarding was used to supply fuel for London. Wood pollards were also important features for farming communities and were often located close to buildings or on rough grazing common land belonging to a village[17].

Pannage

Since Roman times, pigs were released into beech and oak woodlands to feed on the acorn and beech mast, and into fruit orchards to eat fallen fruit. During the Middle Ages, most forests in central Europe were valued by surveyors according to the number of pigs that could be supported by the acorn and beech mast, with woods designated as one-hog to hundred-hog woods[18]. Pannage was a legal term from Norman times describing the right to release swine into a woodland during a specified season[19]. It often involved long-distance movement (droves) of pigs from areas around larger forest complexes. For example, in 17th-century Denmark, animals were driven from the treeless western Jylland to eastern forests[20]. Although pannage in a legal sense had mostly disappeared by the 1800s, it continues today in the New Forest, southern England. In addition to fattening up for the winter, pigs provide a useful service in reducing the density of acorns which are poisonous to the cattle and ponies that graze in the forests[21].

Figure 1. *Dehesa* system in Spain (© ORC 2010).

Dehesas *(Spain)*/montados *(Portugal)*

Dehesa/montado is a Mediterranean land-use system based on widely spaced oaks (*Quercus rotundifolia, Quercus ilex, Quercus pyrenaica* and *Quercus suber*) on shallow stony soils, which produce acorns, wood, charcoal and cork, with sheep, goats, pigs and cattle grazing and browsing beneath the trees (Fig. 1)[22,23]. The system developed primarily to produce fine hams; scattered oaks produce substantially more acorns than woodland trees, which, combined with understorey grazing of grasses and herbs, produce high-quality pigs[6]. The production of cork from the bark of *Q. suber* L. in these systems is of international importance; about 340,000 t of cork are produced annually with Portugal (54%) and Spain (26%) producing most of this[24].

Erosion control and shelter belts

The value of tree planting for controlling soil erosion was recognized back in the 19th century, when the French forestry department planted pine trees on overgrazed, steep slopes[9]. Some traditional European landscapes are shaped by the establishment of shelterbelts centuries ago, such as in the Rhône valley where trees were planted to protect against the mistral winds. In the 1950s and 1960s in the '*bocage*' regions of Normandy and Brittany, the removal of hedgerows led to severe wind erosion[9].

Hedgerows

Traditional hedgerows provided many benefits; in addition to the provision of shelter, hedges provided stock-proof barriers, forage and browse for livestock, as well as food and medicinal plants for rural populations. During the second half of the 20th century, hedgerows were removed to create larger fields for more efficient use of farm machinery. Within the UK, over 50% of hedgerows have disappeared since 1947[25]. Despite recent agri-environment schemes that have helped to slow this decline by introducing options that encourage hedgerow management, re-creation and restoration[26,27], the Countryside Survey reported that the length of 'managed' hedgerows decreased by 6.2% between 1998 and 2007, primarily due to lack of management causing hedgerows to become neglected[28].

Timber tree systems

Intercropping poplar with cereals became fashionable in France in the 18th century, and still covers about 6000 ha in well-irrigated alluvial regions[6]. In the province of Noord-Brabant, in The Netherlands, poplar was integrated with livestock over an area of 3000 ha, for the production of veneer for matchsticks[29].

The Decline of Traditional Agroforestry

Eichhorn et al.[6] identify seven basic causes responsible for the decline of traditional agroforestry systems in Europe:

- Increasing mechanization leading to the removal of scattered trees to facilitate cultivations.
- The post-war demand for increased productivity through monocultures.
- A reduction in the agricultural work force, prohibiting labor-intensive systems such as full-stature fruit orchards.
- A shift from small, fragmented land holdings to larger single farms, with an increase in field sizes, the removal of boundary trees and landscape simplification.
- Policy regimes that favored single crop systems over crop associations.
- For many years, wooded areas were ineligible for subsidy payments, and so trees were removed to maximize subsidy income.
- Stricter quality regulations for dessert fruit leading to intensification of orchard production[6].

Developing Modern Agroforestry

Agroforestry was reintroduced as a modern concept in the late 1970s[30]; since then the emphasis has been on the development of new systems designed to fulfil the potential benefits of increased productivity balanced with resource and environmental conservation. Four main features of traditional agroforestry systems can be embraced in the development of new systems to achieve these aims: resource conservation, energy conservation, multifunctionality and diversity.

Resource conservation

Traditional systems harnessed the efficiency of trees to ensure conservation of the key resources, soil, nutrients, water and biodiversity, and avoided the need for external

inputs to maintain productivity. This was of particular value on marginal land where soil type, climate and/or topography limited the adoption of more intensive forms of agriculture, such as in the Mediterranean *dehesas*. Modern systems can build on this knowledge both to restore degraded land, for example, in areas prone to soil erosion or salinization, as well as to increase the productivity of marginal land[31,32]. By reducing the need for external inputs such as fertilizer and pesticides, these systems should be of particular relevance to low-input and organic farmers. Furthermore, given the acute problem of greenhouse gas emissions and global climate change, it is clear that the incorporation of trees can also provide a major boost to carbon sequestration, conservation and substitution[33–35].

Energy conservation

Trees were once valued for providing shelter for animals, crops and humans, as well as a source of wood fuel; this is becoming particularly relevant once again as we look to conserve energy and develop renewable energy resources. Establishing shelter belts around domestic and livestock housing can reduce energy use for heating or cooling by 30%[36], while providing tree cover on livestock range can reduce the energy needed for regulating body temperatures, and so result in higher feed conversion and weight gain[37]. Traditional agroforestry systems such as wood pollarding provided fuel as its primary woody product. New systems incorporating biomass crops such as short-rotation willow coppice could meet the energy needs beyond the farmgate, while other systems could provide wood fuel for domestic wood burners as a by-product of tree management practices including thinning and pruning operations.

Multifunctional

Trees are multifunctional; in addition to primary outputs of food, fuel and timber, trees can provide a range of other resources and functions for use on farms, such as wood products (e.g., fence posts), seasonal fodder production during shortages of forage, stockproof barriers and wood fuel, as well as protecting the environment[4,5]. Traditional systems exploited the full range of functions to maximize total productivity and minimize inputs; novel systems should look beyond the primary products to identify and exploit these additional benefits through careful design and management This may involve more complex systems of management than currently employed, for example to take advantage of seasonal resources such as acorns and beech mast by running pigs through farm woodlands.

Diverse systems

In contrast to current mainstream agriculture with its focus on a limited range of crops, diversity was inherent to traditional agroforestry systems. This diversity had both economic and ecological benefits; a diversity of products gave both short-term (from crops, livestock, fruits/nuts and wood fuel) and long-term (timber) returns, provided income stability and diversified local economies, while creating a structurally and temporally diverse agroecosystem which benefited associated biodiversity and ecosystem services. It is now becoming generally accepted that, even if the processes are highly complex, there is a general positive relationship in ecosystems between diversity and stability and also between diversity and productivity[38,39]. Modern agroforestry should aim to incorporate diversity within the planned system, both within the tree and crop components to maximize these economic and ecological benefits.

Modern systems in practice

Agroforestry can be established for the production of food, fuel and timber, and for environmental protection such as riparian (riverside) buffers, shelterbelts and soil protection. Examples of modern agroforestry can be found throughout temperate regions of the world.

In North America, there are many current farm practices that fall under the description of agroforestry. These include windbreak systems (shelterbelts), silvopastoral systems, alley–cropping systems, integrated riparian management systems and forest farming[40–43]. Silvopasture is the most common agroforestry practice in the southern United States, most of which is pine-based (*Pinus* spp.)[44]. Much of the research and establishment has utilized black walnut as the tree species of choice as this is the most valuable wood grown in North America[40].

In New Zealand and temperate regions of Australia, agroforestry systems have been developed over the past 30 years to address the problems of land degradation including salinization and soil erosion[45]. Agroforestry systems in Australia include scattered trees in pastures, tree belts and woodlots[45]. In New Zealand, systems that integrate the production of high-grade *Pinus radiata* sawlogs with cattle and sheep grazing the understorey to reduce fire risk and provide early returns from the land have been established in degraded areas to minimize soil erosion[9,46].

Modern agroforestry systems in China have been developed to address three serious issues—environmental degradation, population growth and resource depletion[47,48]. Farmland is limited, with an average of 0.1 ha of land available to a farmer, so agroforestry is seen as a means of maximizing productivity from limited resources as well as bringing marginal land into production[47]. Agroforestry systems are implemented both at the small scale on small land parcels by individual farmers, and at the large scale through government projects. One such project is the 'Three North' project in north-eastern China which aimed to provide environmental services through afforestation using long- and

Figure 2. Intercropped willow coppice system at Wakelyns Agroforestry, England (© Martin Wolfe 2010).

short-term combinations of multi purpose, multiple-storey species on an area covering 4.1 million km^2.[47]

In Europe, Mosquera-Losada et al.[49] identified six basic types of agroforestry: silvoarable, silvopasture, forest farming, riparian buffers, improved fallow and multipurpose trees. Several novel agroforestry systems have recently been developed to investigate the potential of agroforestry to address recent issues such as renewable energy supplies (Fig. 2). A 'combined food and energy' system that integrated bioenergy production from belts of alder, willow and hazel with crop and pasture production has been established in Taastrup, Denmark in 1995[50,51]. At Aarhus University, Denmark, pigs have been released into energy crop systems to provide benefits for both the pigs and crops[52].

Designing and managing agroforestry for maximum, sustainable, production

Interactions between woody and non-woody components in agroforestry can be positive, negative or neutral, and the productivity of a system is a net result of these interactions[53]. Agroforestry systems should be designed to optimize resource capture by maximizing positive interactions and minimizing negative ones. Appropriate selection of the woody and crop or livestock species of the system to meet site and farm business requirements is necessary, as well as careful consideration of the potential interactions between the different species[54]. Ideal tree species for agroforestry systems should maximize niche differentiation between the tree and crop; deep roots are keys to access nutrients and water unavailable to the crop and either a crown that is in leaf outside the crop's main growing period or that casts a light, even shade. The spatial design of the system will also influence productivity by determining the zone of interactions between the trees and crops, and therefore, the relative potential benefits. For example, trees distributed evenly will have a larger zone of interaction with the adjacent crop or pasture compared to a clumped distribution[54] and in temperate regions, orientating tree rows in a north–south direction is generally accepted as the most efficient orientation to optimize direct sunlight penetration to the crop/pasture.

Within agroforestry systems, the productivity of each component can be manipulated by management practices including pruning, weed control and protection from animal damage[55,56]. Controlling the density of the tree canopy through pruning will determine the amount of sunlight reaching the crop or pasture, and is particularly important in hardwood systems to ensure good quality timber. Below-ground pruning of tree roots through management practices such as trenching, knifing, disking or subsoiling aims to minimize belowground competition and so prolong profitable crop production[57]. Weed control is important in the early years after tree planting to reduce competition, and plastic mulching is often used to reduce weed pressure on newly planted trees[58]. Alternatively, ground cover plants such as clover and lucerne can be established within the tree rows to prevent weed invasion, as long as they have minimum competitive pressure on the trees[59].

Future Developments to Optimize Sustainable Intensification

Recent agroecological advances in sustainable production systems can be adopted to optimize temperate agroforestry systems. Composite cross populations (CCP) of cereal and vegetable crops have been created as an alternative to pedigree lines, by the recombination of diverse seed stocks through hybridization[53,60]. This results in a highly heterogeneous crop that provides a dynamic gene pool with the potential to adapt to the local cropping environment and be more resilient to a changing climate. In an agroforestry setting, the use of a CCP may support the development of a crop that is well adapted to the unique abiotic (e.g., shade and water stress) and biotic (competition and allelopathy) conditions within the system (Fig. 3).

Livestock breeding programs over the past 60 years have focused on the genetic improvement of production traits, such as milk yield, growth rate and number of eggs. This has led to specialized high productivity breeds reliant on high nutrient and energy input systems based on feeding of concentrates that could have been used by humans[54]. Concerns about associated negative impacts on animal welfare and the environment have led to new breeding programs (e.g., LowInputBreeds: http://www.lowinputbreeds.org) that aim to improve animal health and efficiencies of organic and low-input production systems by reducing concentrate rations and increasing the use of on-farm forage and fodder. This research will be useful in optimizing livestock production within an agroforestry system, by informing selection of breeds that can effectively use a variety of forage and fodder resources.

Figure 3. CCP of winter wheat in a mixed hardwood system at Wakelyns Agroforestry, England (© Martin Wolfe 2010).

The use of green manures and nitrogen-fixing legumes is an essential part of organic farming systems, and research is increasing our understanding of the dynamics and processes of nutrient cycling, release and availability for the subsequent crops[55]. Many tropical agroforestry systems include leguminous shrubs and trees as a green manure resource, but this practice is less developed in temperate systems. Relatively few of the 650 woody species that are able to fix atmospheric nitrogen occur in temperate regions; of these *Robinia, Prosopis, Alnus* and *Eleagnus* have been investigated for their nitrogen-fixing potential in agroforestry systems[56]. Significant transfer of fixed nitrogen to crops has been observed in a study that showed that 32–58% of the total nitrogen in alley-cropped maize came from nitrogen fixed by the adjacent *Alnus rubra* (Seiter et al. 1995, in[56]). Incorporating these species into an agroforestry design may provide a continuous source of nitrogen that reduces the need for a fertility-building ley within the crop rotation.

It is essential to use tree material with good provenance when establishing a perennial system with trees, but tree breeding programs have so far focused on a limited range of species and good seed stocks for many species are difficult to source. Programs such as the British and Irish Hardwood Improvement Program (http://www.bihip.org) have made some progress toward improving the quality of home-grown hardwoods. However, it is now necessary to also consider the implications of climate change for tree growth and productivity. Modeling of species responses to changing environmental conditions predicts a poleward and upward shift in ranges for many species[57,58] and suggested adaptation strategies include assisted migration that establishes species into areas predicted to be their future habitats[59]. This advocates careful selection of tree stock provenances, with a consideration of including stock from the southern part of the species' range in the northern hemisphere.

Ecological theories such as island biogeography and metapopulation dynamics lend support to a landscape-scale approach to enhancing and supporting biodiversity and have resulted in the development and implementation of a range of agri-environment scheme prescriptions[61]. A heterogeneous farm landscape, with a variety of habitats and structural diversity of vegetation, provides food, nesting and overwintering resources for farmland inhabitants[62]. Additionally, diverse farmland can support biodiversity in otherwise open landscapes and allow movement of species between isolated patches of natural habitats, as well as buffer protected areas from the impacts of more intensive systems[63]. Agroforestry systems should be established and managed taking these principles into account by designing a system that provides a range of habitats and resources and acts as a corridor between natural habitat patches.

Barriers to Wider Adoption

Reisner et al.[64] used a modeling approach to identify the potential for silvoarable agroforestry within 32 European countries and concluded that one of five commercial tree species (*Prunus avium, Juglans* sp., *Populus* sp., *Pinus pinea* and *Q. ilex*) could grow productively in an agroforestry system on 56% of utilized arable land, while providing ecosystem services such as reducing soil erosion and N leaching on 6 and 30 million ha respectively. However, this potential has not yet been fully realized in temperate regions. Three key areas of activity essential for promoting agroforestry into the mainstream are research, dissemination of information and policy.

Research

Scientific research on agroforestry systems started in the late 1970s, and focused on tropical systems; studies on temperate systems only starting to appear in the literature from the early 1990s[65,66]. The long time scale needed for such research is a limiting factor, with very few examples yet available of complete cycles of the systems through to tree harvest. The complexity of the system requires a multidisciplinary approach to research, across a range of temporal and spatial scales. Research needs range from studies at the fine-scale (species interactions), the farm-scale (economic as well as environmental benefits) right up to the landscape-scale (e.g., watershed impacts on nitrate leaching and biodiversity enhancement), national-scale (e.g., home-grown timber and fuel to reduce imports and increase renewable energy production) and global-scale (climate change mitigation and adaptation).

Dissemination

Another primary barrier to wider adoption of agroforestry is limited awareness among farmers and landowners of agroforestry practices and this has been identified by

a number of studies[67,68] (Thomas and Willis, 1997, in[69]). For example, in a recent study only 33% of farmers correctly defined agroforestry as the integration of trees with crops or livestock systems[68]. These studies showed, however, that where farmers were shown agroforestry systems, their level of interest increased. In a survey of farmers in seven European countries, once the concept of agroforestry was explained and demonstrated with photos, half of all farmers interviewed would be interested in attempting silvoarable agroforestry on their farm[68]. This varied according to region: from 18 to 20% in Bedfordshire (UK), Centre (Fr) and Franche Comté (Fr) to 90% in northern Italy. Perceived benefits and constraints of agroforestry also differed regionally. In northern Europe, the principal benefit was considered to be environmental, while in Mediterranean Europe, increased profitability was perceived to be the main benefit. The complexity of work, and mechanization, were identified as the principal constraints in northern Europe, and crop yield decline as the main constraint in the Mediterranean[68]. Other studies indicate that a lack of financial incentives also limits agroforestry adoption (Willis et al., 1993 and Bullock et al., 1994, in[69]). For agroforestry to be adopted on a wider scale, economic viability and practical management skills need to be demonstrated to farmers and landowners. This relies crucially on effective dissemination and therefore outreach support and extension projects are essential[70].

Policy

A lack of policy support is seen as one of the main barriers to wider adoption of agroforestry. The integration of trees at a low density into agricultural land challenges the conventional specialization of forestry and agricultural policy mechanisms[71]. Within Europe, agricultural support is provided through direct aids and market support (Pillar 1) and Rural Development Policy (RDP) (Pillar 2) including forestry policy schemes for farm woodlands. Within Pillar 1, direct payments are made based on the area of eligible agricultural land. Agroforestry is not currently recognized as a valid land use class, and eligibility of a particular agroforestry system depends to a great extent on the nature of the woody component in the system. Timber or wood fuel trees are ineligible for payments unless the area can be grazed or agricultural activities can carry on in the same way as if trees were not present, while systems that include permanent crops such as top fruit, hardy perennial soft fruits such as blackberries and raspberries, nuts and vines are eligible for payments, as are short rotation coppice systems.

Under Pillar 2, agri-environment schemes are designed to protect the natural and historic environment, promote public access and protect natural resources. While agroforestry systems provide a means of improving ecosystem service delivery on farmland, the management needed to maintain productivity often conflicts with management requirements specified by the schemes' traditional agroforestry methods. For example, parklands, hedgerows, wood pastures and traditional orchards are particularly valued for their cultural heritage, and several options in RDP support the restoration and maintenance of these systems. However, within these options there is little emphasis on managing these systems for productivity.

Also under Pillar 2, there is direct support available for agroforestry through Article 44, which supports the first establishment of agroforestry. In 2009, a review of implementation of forestry measures under the RDP by member states found that only 17 Regions or States had adopted this measure[72] although France has recently incorporated Article 44 into its new RDP 'Objectif Terres 2020'. Raising awareness of the potential of agroforestry among policy-makers essential for promoting agroforestry as a mainstream land-use system.

Conclusion

In temperate systems, the general belief seems to be that the high cost of manual labor in Europe necessitates a greater reliance on agrochemical input and intensive management, particularly in the industrialized northern countries. However, we can develop sustainable agroforestry systems by building on successful aspects of traditional systems and assimilating these with recent advances in agroecological approaches to production. This should allow the full potential of agroforestry as a low-input, biodiverse approach to sustainable production and ecosystem service delivery to be realized.

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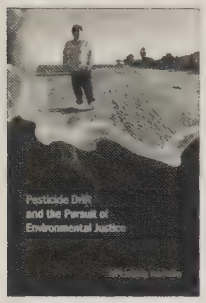

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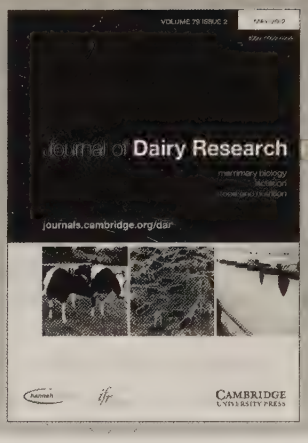

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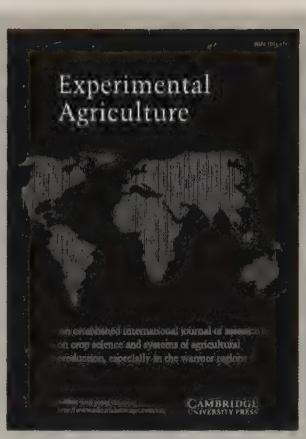

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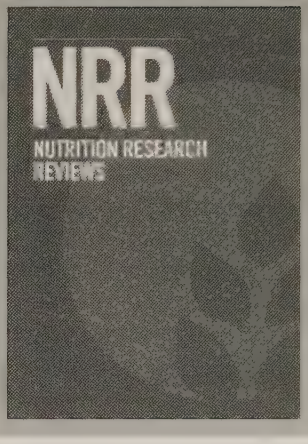

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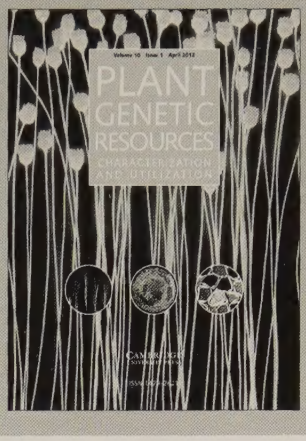

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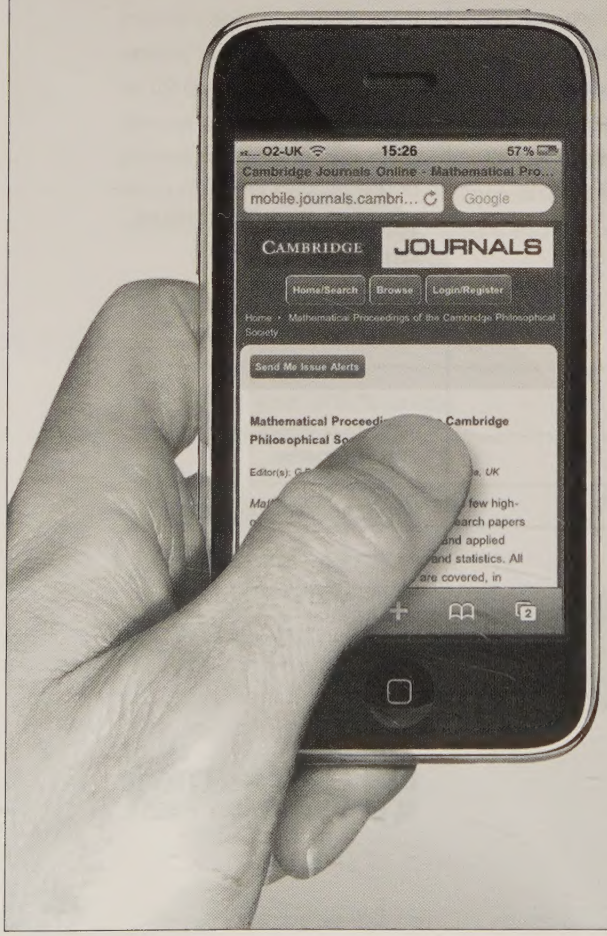

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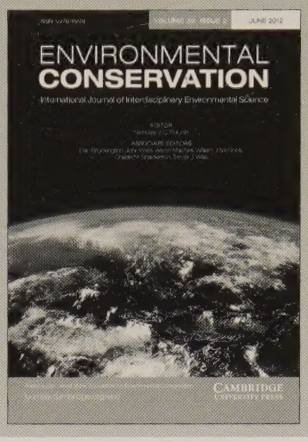

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